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SIMULATION OF LEACHATE GENERATION FROM MUNICIPAL SOLID WASTE

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The simulation of leachate generation from a municipal solid waste landfill or landfill simulation should utilize a mechanistic model which properly accounts for the microbially mediated processes of landfill stabilization. In the past, several leachate generation models have been developed based on the solubility of the waste constitutents in water percolating through a landfill. These models, called "washout models" provided a reasonable approximation of leachate constituent concentrations after the landfill or landfill simulation reached a period of relative dormancy, called maturation, but were deficient in predicting leachate constitutent concentrations in the early stages of the landfill. These early stages in the life of a landfill are extremely important because, in most cases, the highest leachate concentrations and the most extreme conditions a liner or the surrounding environment would be subjected to occur very early in the life of the landfill.

A mechanistic three step model, GTLEACH-I, was developed to simulate the microbially mediated processes of landfill stabilization in terms of hydrolysis of substrate, acid formation and methane fermentation. The model was applied to two sets of experimental data and provided a reasonable prediction of volatile acids and gas generation.

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FOREWORD

Today's rapidly developing and changing technologies and industrial products and practices frequently carry with them the increased generation of solid and hazardous wastes. These materials, if improperly dealt with, can threaten both public health and the environment. Abandoned waste sites and accidental releases of toxic and hazardous substances to the environment also have important environmental and public health implications. The Hazardous Waste Engineering Research Laboratory assists in providing an authoritative and defensible engineering basis assessing and solving these problems. Its products support the policies, programs and regulations of the Environmental Protection Agency, the permitting and other responsibilities of State and local government and the needs of both large and small businesses in handling their wastes responsibly and economically.

This document discusses the mechanisms of microbially mediated landfill stabilization and the resulting impact on leachate generation. Several hydrologic and biological models are reviewed and a new model, GTLEACH-I, proposed. The biological module of GTLEACH-I is a mechanistic three-step model which extends a two-step model proposed by Straub and Lynch (1982). The fit of the model to two sets of experimental data is discussed.

The intended audience for this document includes those involved in analytical or numerical simulation of landfill stabilization processes.

Thomas R. Hauser
Director
Hazardous Waste Engineering
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ABSTRACT

The simulation of leachate generation from a municipal solid waste landfill or landfill simulation should utilize a mechanistic model which properly accounts for the microbially mediated processes of landfill stabilization. In the past, several leachate generation models have been developed based on the solubility of the waste constituents in water percolating through a landfill. These models, called "washout models" provided a reasonable approximation of leachate constituent concentrations after the landfill or landfill simulation reached a period of relative dormancy, called maturation, but were deficient in predicting leachate constituent concentrations in the early stages of the landfill. hese early stages in the life of a landfill are extremely important because, in most cases, the highest leachate strengths and the most extreme conditions a liner or the surrounding environment would be subjected to occur very early in the life of the landfill.

A mechanistic three-step model, GTLEACH-I, was developed to simulate the microbially-mediated processes of landfill stabilization in terms of hydrolysis of substrate, acid formation and methane fermentation. The model was applied to two sets of experimental data and provided a reasonable prediction of volatile acids and gas generation.

This report was submitted in fulfillment of Cooperative Agreement CR812580010 by the Georgia Institute of Technology under the sponsorship of the U.S. Environmental Protection Agency. This report covers the period for September 1985 to March 1987, when the work was completed.

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SECTION 1

INTRODUCTION

The generation of leachate from a landfill is a complex process depending not only on the characteristics of the landfilled wastes, but also on the interaction of the waste with water percolating through the landfill, operational variables such as waste placement conditions, climatic conditions, the physical design of the landfill, and the interaction of the landfilled waste with ground water, as discussed in Appendix A. Leachate characterization is further complicated by the effects of microbial activity, which mediates the fate of both hazardous and nonhazardous waste and their potential for migration from the landfill site.

Leachate characteristics and the rate of leachate generation are dependent on time and stage of landfill stabilization. Brunner (1979) investigated the characteristics of leachate generated from municipal waste disposal and found that the highest concentrations of a large number of constituents were generated during the early stages of microbially mediated stabilization. Therefore, if the purpose of any model is to predict changes in chemical concentrations of leachate for assessment of migration potential or liner/leachate compatibility, it is necessary to model the various phases of landfill stabilization.

LANDFILL STABILIZATION

The fate of waste constituents disposed in a landfill can be envisioned as a partitioning among the solid, vapor or aqueous phases. Various microbial, chemical, and physical transformations alter the chemical and physical nature of the waste and transfer it from one phase to another. Likewise, waste constituents are transported out of the landfill system in aqueous solution and suspension by washout of leachate and through the evolution of gases.

Microbially-mediated reactions control the landfill environment for some time after initial placement of waste and strongly impact the outcome of other chemical and physical transformations leading to stabilization. Both aerobic and anaerobic microbial processes occur in a landfill, however, free oxygen availability is typically limited to early stages of the landfill stabilization, and is often exhausted prior to the appearance of leachate. For this reason, anaerobic activity usually establishes and controls leachate quality during the active life of a landfill.

Anaerobic processes are often described as multistep in nature. The first step, hydrolysis of complex substrate, results in conversion of solid organic matter to soluble compounds and reduction in the size of organic molecules to facilitate transport across microbial cell membranes. The second step, acidogenesis, is described as one in which acid-producing bacteria utilize hydrolyzed organic molecules as carbon and energy sources, producing an abundance of short-chain volatile acids as conversion products. Carbon dioxide and hydrogen gas may also be evolved during this step. The products of acidogenesis (volatile acids, CO_2 and H_2) are converted to methane gas by methanogenic bacteria in the final anaerobic process step.

In a steady-state anaerobic treatment process, the three steps occur simultaneously at a rate controlled by the slowest step on the sequence so that there is little accumulation of intermediate products over time. However, in a non-homogeneous, batch-wise system such as a landfill, the activity of acid-forming and methane-forming bacteria may not be in balance at any particular location in the landfill at any one time. As a result, landfills are often characterized by temporal stages caused by the predominance of different microbial populations at different times in the life of the landfill. Pohland, et al., (1985) described five phases through which a landfill proceeds as it becomes stabilized; initial adjustment, transition, acid formation, methane fermentation and final maturation. These phases can be outlined as follows:

Phase I: Initial Adjustment--

Initial waste placement and preliminary moisture accumulation.

Initial subsidence and closure of each landfill area.

Changes in environmental parameters are first detected to reflect the onset of stabilization processes which are trending in a logical fashion.

Phase II: Transition--

Field capacity is approached and leachate is formed.

A transition from initial aerobic to facultative anaerobic microbial stabilization occurs.

The primary electron acceptor shifts from oxygen to nitrates and sulfates with the displacement of oxygen by carbon dioxide in the gas.

A trend toward reducing conditions is established.

Measurable intermediates such as the volatile acids first appear in the leachate.

Phase III: Acid Formation--

Intermediary volatile acids become predominant with the continuing hydrolysis and fermentation of waste and leachate constituents.

A precipitous decrease in pH occurs with a concomitant mobilization and possible complexation of metal species.

Nutrients such as nitrogen and phosphorus are released and utilized in support of the growth of biomass commensurate with the prevailing substrate conversion rates.

Hydrogen gas may be detected and may affect the nature and type of intermediary products being formed.

Phase IV: Methane Fermentation--

Intermediary products appearing during the acid formation phase are converted to methane and excess carbon dioxide.

The pH is elevated from a buffer level controlled by the volatile acids to one characteristic of the bicarbonate buffering system.

Oxidation-reduction potentials are at their lowest values; sulfates and nitrates have been reduced to sulfides and ammonia.

Nutrients continue to be consumed.

Complexation and precipitation of metal species with sulfides and organic ligands proceed.

Leachate organic strength is dramatically decreased in correspondence with increases in gas production.

Phase V: Final Maturation--

Relative dormancy following active biological stabilization of the readily available organic constituents in the waste and leachate.

Nutrients may become limiting.

Measurable gas production all but ceases.

Typical subsurface environmental conditions become reinstated.

Oxygen and oxidized species may slowly reappear with a corresponding increase in oxidation-reduction potential.

More microbially resistant organic materials may continue to be slowly degraded with the possible production of humic-like substances capable of complexing and re-mobilizing heavy metals.

Thus, microbially-mediated reactions accomplish first the transformation and solubilization of waste material into the aqueous solution phase. Further transformation into volatile acids followed by conversion to methane result in transfer of conversion products to the vapor phase. The relative rates of these transformations combined with the rate of moisture infiltration determine the concentration and mass flux of the various biochemical intermediates and products in the leachate.

Superimposed upon the microbial transformations are solubility, speciation, oxidation-reduction and vapor-liquid equilibria of both inorganic and organic chemicals. These equilibria result in partitioning of chemical components among solid, aqueous and vapor phases. Changes in these equilibria as landfill stabilization progresses are evidenced by the decrease in pH during the acid formation phase and subsequent increase in pH as methane fermentation is established, mobilization of metals during the acid formation phase and then precipitation principally with sulfides during methane fermentation, and changes in oxidation-reduction potential from positive to negative.

Sorption, the partitioning of organic and inorganic chemicals between solution and solid surfaces, is a phenomenon not well understood in a landfill setting. Many factors can affect the sorption of a substrate, such as:

- temperature;
- pH and ionic strength for substances that ionize;
- the presence of other sorptive substances which can compete for the available sites; and,
- particle size distribution and available surface area.

With regard to adsorption, current prediction is largely limited to single solute systems and is based on experimentally determined isotherms. Most information concerning adsorption in natural systems is limited to soil surfaces, and little is known about the adsorptive capacity of municipal solid waste.

One final transformation to be considered, particularly for toxic organic compounds which may be microbially degraded at relatively slow rates, is chemical degradation. Although the mechanism for chemical degradation depends on the specific organic chemical and environmental conditions, in an aqueous environment away from light and oxygen, hydrolysis is typically the dominant mechanism (Verschuren, 1983). Chemical-specific hydrolysis rate constants are needed to predict the chemical degradation of organic chemicals in the landfill environment.

One of the major difficulties in describing the fate of waste disposed in a landfill stems from uncertainties in the relative impacts of the various transformation and partitioning processes acting on the waste, leachate and gas. Leachate and gas composition and quantity data reveal the net results of the various transformation and partitioning processes, but do not necessarily indicate the path taken. Furthermore, data are seldom available and sufficient to describe the composition of the solid phase. Thus, it is difficult to assess the relative impacts of adsorption, complexation and precipitation processes, since they all result in transfer of a component from the aqueous to the solid phase or vice versa. Likewise, assessment of the relative effects of microbial degradation, chemical conversion and sorption on the fate of organic components disposed in a landfill is also difficult.

SECTION 2

CONCLUSIONS

A useful interpretation of leachate generation and leachate characteristics is a highly complex and difficult undertaking, primarily because landfill disposal of waste is site-specific. The local hydrogeology and landfill design and operation all influence the potential for infiltration and percolation and the rate of saturation of the waste materials. Therefore, leachate characteristics and opportunity for migration will vary in accordance with these factors.

Numerous evaluations and simulations of landfill performance have been conducted to characterize leachate generation patterns. Some of the data from these studies have been used to develop numerical predictive models descriptive of leachate generation as a function of time. Based on a review and further development of these models, the following conclusions are presented:

1) Data descriptive of microbial mediation processes of landfill stabilization are necessary components of an effective numerical landfill simulation model and are influenced by waste type, availability of nutrients, moisture content, and biological or physical/chemical conversion.

2) Comprehensive analysis of waste characteristics and waste constituent distribution must be available as input data for predicting leachate constituent concentration changes as a function of time, to indicate substrates susceptible to conversion, to help assess any delays in the progress of landfill stabilization, and to appropriately distinguish microbial mediation from simple washout.

3) There are no established techniques to yield data sufficient to accurately predict hydraulic conductivity at landfills. Data from simulation studies performed to date are presumptive of flow conditions and generally ignore non-homogeneity of the leaching matrix.

4) The GTLEACH-I model has been developed to simulate the microbially mediated processes of landfill stabilization in terms of hydrolysis of waste substrate, acid formation and methane fermentation.

5) GTLEACH-I provides reasonable prediction of volatile acids and gas generation during landfill stabilization and may be expanded to

predict variations of pH, pE, and ORP as functions of time. These latter relationships again require a comprehensive database which must include acid-base and oxidation-reduction equilibria not usually present in compilations of existing landfill analyses.

6) Calibration of the GTLEACH-I model has been limited by a general lack of substrate specificity, the uncertainty of flow distribution and short-circuiting particularly during the early stages of landfill stabilization, the possibility of retardation or inhibition of microbial mediation, the influences of population dynamics as substrate conversion proceeds and engulfs the waste mass, and the potential for containment or release of the liquid and gas transport phases.

SECTION 3

RECOMMENDATIONS

Based on the insights obtained from the review and utilization of available data, the following recommendations for future research are proposed:

1) Develop implementation strategies for any future research on landfill stabilization to assure that sufficient data are generated to support the further development and verification of predictive models capable of accurately predicting both short-term and long-term variations in leachate quality and quantity.

2) Initiate a field study which provides sufficient data to support the refinement and verification of the numerical model.

3) Initiate complementary studies to measure hydraulic conductivity of landfilled wastes to variable physical and chemical characteristics in a temporally and spatially distributed fashion. The study should provide data and information regarding spatial variation of hydraulic conductivity and time related changes in hydraulic conductivity.

4) Refine and expand the GTLEACH-I model to predict pH, pE, and ORP as a function of time, to incorporate more realistic descriptions of flow retardation or inhibition, and to allow prediction of selected inorganic and organic constituent concentrations as population dynamics affect the overall processes of landfill stabilization.

SECTION 4

REVIEW OF PREVIOUS MECHANISTIC MODELS OF LANDFILL STABILIZATION

A brief review of previous work on mechanistic models of leachate quantity and quality during stabilization of municipal solid waste landfills follows. The purpose of this review is to examine previous approaches and assumptions in modeling landfill stabilization and to assess the degree of success achieved in simulating experimental data.

The transfer of solid organic matter into a soluble form was assumed by Straub and Lynch (1982) to follow a pseudo first-order kinetic reaction of the form:

$$R_g = \frac{S^m}{S_o} b (C_{max} - C) \qquad (1)$$

where;

R_g = net rate of transfer of COD from solids to landfill moisture,

S = mass of solid COD/unit bulk volume of landfill,

S_o = initial or ultimate S,

m = dimensionless constant,

b = solubilization rate constant,

C_{max} = saturation COD concentration in leachate, and

C = COD concentration in landfill moisture.

Thus, solubilization of solid substrates was treated as an equilibrium process in which the rate of mass transfer decreased as the initial source was depleted. Microbially-mediated aspects of solubilization were not included in Equation 1 implicitly.

Organic substrate (COD) utilization or removal by suspended microorganisms was described as following the Monod kinetic model:

$$R_u = \frac{-k_o X S}{Y(k_m + S)} \tag{2}$$

where;

R_u = substrate utilization rate,

S = substrate concentration in leachate,

X = microbial cell concentration in leachate,

k_o = specific substrate utilization constant,

k_m = half velocity constant, and

Y = microbial cell growth yield coefficient.

Microbial cell growth rate was then given by:

$$R_c = - Y R_u - k_d X \tag{3}$$

where;

k_d = endogenous death rate of microbial cells, and

R_u and Y were defined previously.

Straub and Lynch (1982) applied the preceding mass transfer equations to three models of moisture and contaminant transport: a single well-mixed reactor model, a vertically cascaded reactors model, and a model of unsaturated flow and contaminant transport through porous media. In the first model, a landfill was considered to be a single well-mixed reactor, while the vertically-cascaded reactors model was composed of a number of these well mixed reactors connected in series. The vertically-cascaded reactors model was thought to model vertical transport processes more realistically than the single, well-mixed reactor. The moisture content of the wastes in both models was assumed to be at field capacity. Flow into the system, flow between reactors and effluent leachate flow were set equal at any instant in time for the vertically-cascaded model.

The model of unsaturated flow and contaminant transport through porous media was based on an explicit finite-difference representation of the vertical transport equations:

$$\frac{\delta\theta}{\delta t} = \frac{-\delta k(\theta)}{\delta z} + \frac{\delta}{\delta z}\left(D(\theta)\,\frac{\delta\theta}{\delta z}\right) - r \qquad (4)$$

and,

$$\frac{\delta C}{\delta t} = -\frac{q}{\theta}\,\frac{\delta C}{\delta z} + \frac{1}{\theta}\,\frac{\delta}{\delta z}\left(E\,\frac{\delta C}{\delta z}\right) + R + \frac{rC}{\theta} \qquad (5)$$

where;

θ = moisture content,

C = contaminant concentration in leachate,

t = time,

z = position (depth) in landfill,

$k(\theta)$ = unsaturated hydraulic conductivity,

$D(\theta)$ = unsaturated moisture diffusivity,

r = moisture sink,

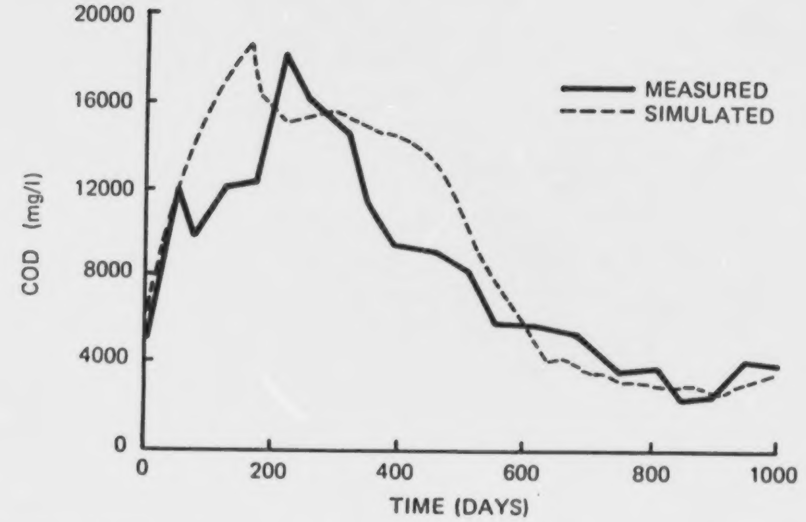
q = moisture flux,

E = contaminant diffusion/dispersion coefficient, and

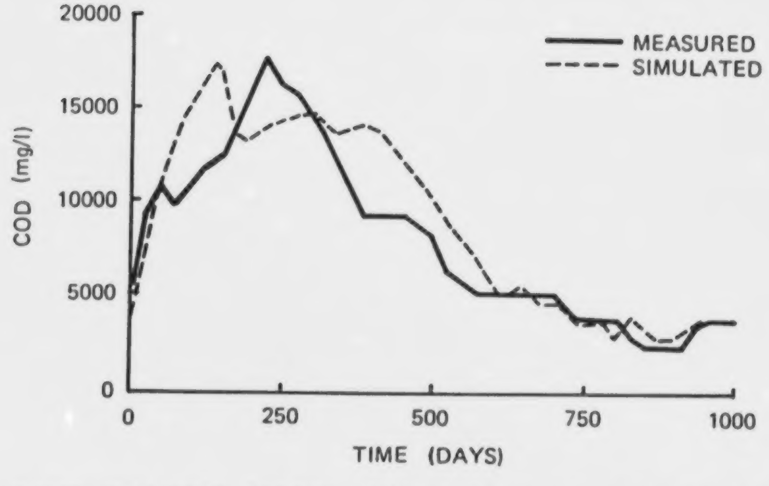
R = net contaminant generation rate.

Straub and Lynch (1982) applied the three transport models to a simulation of a single-pass experimental landfill cell using data reported by Pohland (1975). The ultimate source of leachable organics was estimated as 12% of the dry refuse mass. The value of m was assumed to be 2 as a reflection of the decreasing availability of easily leachable organic material as leaching proceeded, and C_{max} was inferred from maximum COD level typically observed experimentally (40,000 - 50,000 mg/L). The solubilization rate constant and Monod parameter were estimated or adjusted to fit the test data, as were the parameters for the vertical transport equations. The vertically-cascaded reactors model and the model of unsaturated flow and contaminant transport through porous media model exhibited very similar behavior and fit the experimental data reasonably well as shown in Figure 1. The single-reactor model showed somewhat poorer fit than the other two models.

Edrogan (1982) presented two approaches for modeling leachate generation and transport through a sanitary landfill, which were essentially the same as those of Straub and Lynch (1982). The first approach modeled the landfill



UNSATURATED TRANSPORT MODEL SIMULATION OF POHLAND DATA (1975)
(Straub and Lynch, 1982)



FOUR–REACTOR MODEL SIMULATION OF POHLAND DATA (1975)
(Straub and Lynch, 1982)

FIGURE 1. Comparison of Unsaturated Flow Model and Four-Reactor Simulations.

as a number of continuously-stirred tank reactors (CSTR) in series, while the second approach employed the concepts of flow through porous media. Both approaches were applied for inorganic and organic pollutants and allowed for solubilization of organics and inorganic matter by first-order kinetics; removal of organic matter by microbial action of a single group of suspended organisms; and accumulation of biomass by growth. The following assumptions were made in developing the models:

- medium is unsaturated;
- moisture flow is unidirectional and moved vertically downward through the landfill;
- inorganic and organic contaminants are generated from the solid phase by first-order reaction;
- organic substances are utilized by microorganisms as a food and energy source; and,
- the roles of aerobic and anaerobic microorganisms were simulated using Monod kinetic formulation.

The model was not applied to the prediction of experimental data so no conclusion could be drawn regarding its usefulness.

Bernardes (1984) developed a model to describe fixation of heavy metals in the codisposal of industrial sludges with domestic solid waste. Three sub-models were presented: a biological model of the decomposition of domestic solid waste over time; a hydrologic model for unsaturated flow as applied to landfills; and, a chemical equilibrium model to predict metal speciation with age of the landfill. Biodegradation of solid waste was modeled in three steps. First, four major components consisting of cellulose, protein, fat, and starch were hydrolyzed to soluble complex organic matter, measured as COD. Hydrolysis was described as an enzyme-catalyzed reaction which, under substrate to microorganism ratios, was a zero-order kinetic expression:

$$\frac{dM}{dt} = -K\ (V \times \theta) \tag{6}$$

or,

$$M/M_o = 1 - K't \tag{7}$$

where;

t = time (years),

M = mass of solid present (kg COD),

M_o = initial mass of solid present (kg COD),

V = bulk volume of solid waste,

θ = moisture content, and

K' = hydrolysis constant (year $^{-1}$).

Hydrolysis of each substrate was mediated by a specific group of microorganisms, each with a specific hydrolysis constant. Hydrolysis was followed by acidogenic and methanogenic phases. No further explanation of the biological decay model was given.

Outputs from the biological model developed by Bernardes (1984) were: cellulose, protein and fat content in the solid phase as a function of time; glucose, amino acid, propionic acid, butyric acid, hydrogen gas, soluble fat, sulfide and sulfate concentrations in solution as a function of time; and, concentration of five microbial populations (acid-formers metabolizing glucose, acid-formers metabolizing amino acid, methanogens metabolizing acetic acid, methanogens metabolizing propionic acid, methanogens metabolizing hydrogen gas) as a function of time. Inputs to the biological model included: coefficients for each of the enzyme-catalyzed reactions, density of solid waste, initial moisture content, actual moisture content, desired time step, porosity, molar volume of gas, and initial values of total CO_2 plus CH_4 pressure.

Bernardes (1984) employed a computer model developed by McDuff and Morel (1973), REDEQL 2, to evaluate chemical equilibria described by Stumm and Morgan (1981). The user was required to identify all species present, the total concentration of each element present, and either the pH or the total equilibria, correct for ionic strength and temperature, and apply the proton condition to avoid round-off errors at high ionic strengths. REDEQL 2 would then output the fraction of the total concentration of a metal in solution in each form and the fraction which precipitated.

Based on data gathered by Pohland (1980), Bernardes (1984) assumed that:

- over the pH range encountered for leachates, the dominant carbonate species are $H_2CO_3^*$ and HCO_3^-;
- the pH changes slowly enough to assume a series of metastable equilibria;
- an important fraction of the alkalinity is due to acetic acid; and,
- initially the dissolved CO_2 and $H_2CO_3^*$, is in equilibrium with HCO_3^-, but not with CO_2. Not until gas production is well established is the $H_2CO_3^*/CO_2$ equilibrium attained.

Mass balances for HCO_3^- and CO_2 were made taking into account biological sources and gas transport. The pH was calculated from the mass balance and equilibria expressions. The Davies expression (Stumm and Morgan, 1981) for ion activity coefficients was used to correct the equilibrium constants for ionic strength. The pE based on oxidation of glucose to acetic acid and CO_2 to methane was also calculated.

In developing the hydrologic model, Bernardes (1984) assumed that:

- flow through landfills is characterized as unsaturated, therefore, water which infiltrates the landfill moves downward due to capillary action and gravity;

- during periods of moisture inflow, the moisture profile moves downward as a wetting front; and,
- during periods with no moisture inflow, the porous media drains.

The wetting front velocity was defined as:

$$\frac{dz_f}{dt} = \frac{k(\theta_1) - k(\theta_2)}{\theta_1 - \theta_2} \tag{8}$$

where;

dz_f/dt = wetting front velocity,

$k(\theta)$ = hydraulic conductivity, and

θ = moisture content as a function of depth, z.

The moisture velocity for a drainage wave was:

$$\frac{dz}{dt} = \frac{dk}{d\theta} \quad (\text{for } \theta = \text{constant}) \tag{9}$$

An expression to relate unsaturated conductivity to moisture content was also employed:

$$k(\theta) = ks\,\frac{(\theta - \theta_o)^3}{(E - \theta_o)} \tag{10}$$

where;

$k(\theta)$ = unsaturated hydraulic conductivity,

k_s = saturated hydraulic conductivity,

θ_0 = field capacity, and

E = porosity.

In cases with variable infiltration rates, where one front could overlap another, the front with higher moisture velocity dominated.

Bernardes (1984) applied his model to simulate experimental results obtained by Pohland (1979) for pH and pE of leachate; by Swartzbaugh (1979) for metal speciation; and by Fungaroli (1979) and Swartzbaugh (1979) for leachate flow. However, it is difficult to make definitive conclusions as to the success of the model due to the limited number of data points that were presented for comparison.

Review of the important efforts in development of mechanistic models of landfill stabilization revealed two dramatically different approaches to modeling microbial processes occurring in a landfill. On one hand, Straub and Lynch (1982) employed a relatively simplistic model of landfill stabilization involving release of COD through microbial metabolism. At the other extreme, Bernardes (1984) developed a very complicated series of microbial processes involving six groups of microorganisms following Monod kinetics.

In selecting a model, one must consider the database available to support and test the model. A highly sophisticated model is useless without adequate data. The most commonly available leachate organic strength measurements are COD and BOD_5. Less frequently gathered are total volatile acid concentration. Finally, only studies such as those conducted by Pohland and his co-workers (Chang, 1982; Schaffer, 1986 Yari, 1986) include analyses of the individual volatile acids as well as many of the other indicator parameters discussed previously.

The overwhelming requirements for Monod parameters and detailed analyses of the solid waste implicit in the model of Bernardes (1984) are impediments considering the current database available for landfill stabilization. However, the limitation of the model of Straub and Lynch (1982) is that it predicts only COD as a measure of leachate quality and fails to describe the phases of landfill stabilization with important fluctuations in pH and ORP and resulting implications for chemical equilibria. If a model of landfill stabilization is to be eventually applied to the prediction of the rate of hazardous as well as nonhazardous organic and inorganic constituents disposed in a landfill, a model which includes elements such as fluctuations in pE and ORP will be essential. One must also bear in mind, however, that landfills are extremely complicated nonhomogeneous systems of microbial and chemical reactions and equilibria for which an exact model would present a considerable challenge and for which it would be even more difficult to obtain the data necessary for verification.

Based on the preceding review, a model somewhere between the two extremes and capable of utilizing available landfill or pilot-scale experimental data, yet sufficiently mechanistic that it is also capable of simulating the observed trends in landfill stabilization, would be the most desirable type of model to develop. Therefore, it was with this goal in mind that the model described herein, GTLEACH-I was conceived.

SECTION 5

GTLEACH-I MODEL DEVELOPMENT

The GTLEACH-I model is a component model composed of a hydrogeologic module and biologic module. The modules comprising the GTLEACH-I model are described below.

THE HYDROGEOLOGIC MODULE

The rate and quantity of water flowing through waste materials can be correlated to the quantity and, to some extent, the characteristics of leachate produced in landfills. This is not to imply that a simple washout model adequately predicts leachate characteristics. Stated very simply, the characteristics of the leachate depend primarily on the adsorption, solution, movement and hydrolysis/conversion of waste materials described by gravity flow models and the biologically-mediated stabilization processes described by biological waste conversion models.

The latter biologically-mediated stabilization processes are dependent on a large number of parameters, among which are the water content, porosity, and distribution of water in the waste, the contact time between water percolating through the landfill and the waste materials, and changes in waste surface area, composition and porosity with time. These parameters either affect or are dependent upon the quantity and rate of water flowing into and through the waste materials.

The hydrogeologic model must account for all flow into and out of the landfill and its constituents cells. Therefore, it is proposed that a quasi-two dimensional deterministic model, similar to the HELP (Schroeder et al., 1983) model, be incorporated into the hydrogeologic module of GTLEACH-I. The HELP model performs a sequential time-based water budget for a landfill cell. The water budget is based on soil and waste characteristics, climatological data and landfill design parameters. The model can be described as a component, semi-empirical numerical model which evaluates the effects and interaction of runoff, evapotranspiration, percolation and lateral drainage. The algorithms used in the HELP model are described by Schroeder et al. (1983) and are not repeated herein.

In order to provide the data required to evaluated the time-dependent progression of biologically-mediated stabilization, several modifications of the HELP model are required. First, the flow model utilized in the HELP model yields unrealistically high flow rates through the waste materials. Therefore, it was anticipated that the vertical flow model would be replaced with a different flow model which could provide more realistic flow rates and distribution of moisture content as a function of depth and time in the waste material. In addition, surface run-on interaction with the site hydrology and short-circuiting were considered necessary components. The flow models and additional components are described in the following narrative.

Since the flow through the waste may be viewed as a series of reactors where the water interacts with the waste materials over a finite period of time, the mixing regimes within reactors are typically described as falling between two idealized extremes; plug flow and completely-mixed flow. In an idealized plug-flow reactor there is no axial mixing, only radial mixing, so that differential cross-sectional fluid elements moving axially through the reactor are homogeneous or completely mixed in the radial direction (perpendicular to the direction of flow). There is no dispersion or mixing between fluid elements in the axial direction. At the other extreme, an idealized, continuously stirred tank reactor is instantaneously mixed with the contents of the reactor and a fluid element leaving the reactor is of identical composition to the contents of the reactor.

Mixing regimes for real or non-real reactors are typically modeled as modified versions of one of the two idealized models (Levenspiel, 1972). The dispersed plug-flow model and the tanks-in-series model, both single-parameter models, are the two most commonly employed models for non-ideal flow. The dispersed, plug-flow model or dispersion model possesses a plug-flow mixing regime upon which is superimposed some degree of backmixing or intermixing that is independent of position within the reactor. The model assumes no stagnant pockets or gross short-circuiting of the fluid in the vessel. As the mixing intensity, the adjustable parameter, is increased, the mixing regime ranges from plug-flow to completely-mixed flow. Axial mixing of fluid flow is described by an equation analogous to Fick's Law of diffusion:

$$\frac{\delta C}{\delta t} = D \frac{\delta^2 C}{\delta x^2} \qquad (11)$$

where; D = longitudinal or axial dispersion coefficient,
C = constituent concentration,
t = elapsed time, and
x = distance in direction of flow.

According to Levenspiel (1972), the dispersion model satisfactorily represents flow which does not deviate greatly from plug-flow. With increasing axial dispersion, however, it becomes increasingly unlikely that the assumptions of the dispersion model will be satisfied by the real system.

The other widely-applied model on non-ideal flow is the tanks-in-series model. In this model the fluid is viewed as flowing through a series of equal size, continuously-stirred tanks, the adjustable parameter being the number of tanks in series. As the number of tanks is increased, the mixing regime varies from a single, completely-mixed tank to plug-flow. Recycle of effluent can be represented equally well by the dispersion model and the tanks-in-series model. "Which approach one takes is simply a matter of taste, style and mood" (Levenspiel, 1972).

For reactors which cannot be satisfactorily represented by one of the two single-parameter models, other more complicated multi-parameter models may be considered. These models are typically formulated by interconnecting several different regions of plug, mixed, dispersed, and stagnant flow in various ways. For trickle beds, chromatographs and packed beds of porous and/or adsorbing solids, a Guassian residence time distribution curve with an extended tail is observed. This phenomena is explained as hold-up of fluid by adsorption on the solid surface, trapping of fluid within pores, or hold-up of fluid in the many stagnant regions formed at the contact points of the packing within the reactor. In these cases, various multi-parameter models have been applied when the single-parameter models were deemed inadequate. One of the simplest and most effective of these models is a modification of the tanks-in-series model in which each of the tanks is connected to a side tank or deadwater tank. As the fluid moves through each vessel, part of the fluid is held up in the side tank. The three adjustable parameters in this model are the number of stages, the fraction of the total volume which is stagnant, and the ratio of flow between the deadwater tank and the active tank to the flow through the entire stage.

In selecting a flow model for a reactor, one would first select a single-parameter model in order to simplify the procedure of fitting the data to the models. If the fit of neither of the single-parameter models is adequate, then a multi-parameter alternative which can accomplish the satisfactory modeling of the mixing regime should be selected. One further consideration in selection of a flow model for incorporation into a predictive landfill model is the need for flexibility in the flow model to accommodate landfills of varying configuration and composition. This is particularly true for GTLEACH-I which was to be calibrated against pilot-scale landfill data in which the depth of the reactor is typically greater than the width, but was to be applied eventually to full-scale landfills which are of much greater width than depth. In the former, one expects to approach a plug-flow mixing regime whereas in the latter, substantial deviations from plug-flow are expected.

Straub and Lynch (1982) have demonstrated that the dispersed-flow and tanks-in-series models yield very similar results in simulating contaminant transport through pilot-scale landfills. At this time, however, there is a lack of data with which to evaluate the extent of deviation from plug-flow encountered in full-scale landfills. Due to this uncertainty and the resulting unreliability of the dispersed flow model for modeling these systems, the tanks-in-series model was selected over the dispersion model for modeling the mixing regime in GTLEACH-I. In addition, if ever the tanks-in-series model was found to be inadequate when applied to full-scale landfill systems, the tanks-in-series model could be modified to incorporate deadwater tanks at each stage.

The tanks-in-series approach may be modeled using either a one-dimensional saturated or unsaturated flow model. The saturated flow model is based on the Darcy Equation:

$$q = k_{ave} \; i \; A \tag{12}$$

where;

q = the flowrate through the waste,

k_{ave} = the average hydraulic conductivity of the waste,

i = the hydraulic gradient, dh/dl

h = the head difference across a layer,

l = the height of the layer, and

A = the cross-sectional area of flow.

In a larger profile, the average hydraulic conductivity of the media may be approximated using the following equation:

$$k_{ave} = \frac{\sum_{i=1}^{N} di}{\sum_{i=1}^{N} \left(\frac{di}{k_{ave_i}} \right)} \tag{13}$$

where;

di = the thickness of the ith waste layer,

k_{ave} = the hydraulic conductivity of the ith waste layer, and

N = the number of layers of waste.

A FORTRAN program, SAT, based on the above algorithm, was written and used to evaluate the impact of flow rate on biologically mediated stabilization. In addition to the saturated flow model, short-circuiting, prior to reaching field capacity, was modeled assuming a low initial effective porosity and a corresponding low initial cross-sectional area of the flow path. The equation used to evaluate the flow rate through the waste attributed to short-circuiting was evaluated using the following algorithm:

$$q_s = k_{was} \; i \; A' \tag{14}$$

where;

q_s = flow rate of water which percolates through the landfill prior to reaching field capacity,

k_{was} = hydraulic conductivity of the waste,

i = hydraulic gradient which is assumed to be equal to 1 for short-circuiting, and

A' = initial cross-sectional area of the flowpath.

The initial cross-sectional area of the flowpath may be evaluated using the following equation:

$$A' = n_e A \tag{15}$$

where;

A = the cross-sectional area of the landfill or lysimeter, and

n_e = the initial effective porosity of the waste which is likely in the range of 0.01 to 0.1.

A listing of program SAT is presented in Appendix B.

As discussed subsequently, the saturated flow model did not adequately model the flow through the waste materials. Results of the modeling using the biological module of GTLEACH-I indicated that a partially saturated flow model must be used to properly characterize flow through the waste materials.

As stated previously, the flow of water through partially saturated soils may be described using the Darcy Equation. For flow in the vertical direction, the velocity may be calculated as:

$$v_z = -k \frac{\delta\phi}{\delta z} \tag{16}$$

where;

v_z = velocity in the vertical direction,

ϕ = total potential or hydraulic head,

z = distance over which the head is dissipated, and

k = hydraulic conductivity of the porous media.

In a saturated porous media, the hydraulic conductivity, k, is primarily a function of the media structure, void ratio, state of stress, stress history, the boundary condition affecting flow and the properties of the permeant (water). In a partially saturated media, the hydraulic conductivity is also a function of the volumetric water content. Therefore, in a homogeneous, isotropic, partially saturated media, the Darcy Equation becomes:

$$v_z = -k(\theta) \frac{\delta\phi}{\delta z} \qquad (17)$$

The total potential in a partially saturated media consists of a pressure head or matric potential, ψ, and an elevation head, z, assuming the velocity head, $v^2/2g$, can be neglected. For unsaturated flow, the pressure head, ψ, is always less than zero, thereby indicating suction in the media pores. If the interconnected media pores are modeled as capillary tubes, the suction which develops in the pores may be described as surface tension forces developing at the air/water interface of the capillary tube.

The general equation for one-dimensional flow through a partially saturated, rigid porous medium is:

$$\frac{\delta\theta}{\delta t} = - \frac{\delta}{\delta z} (v_z) \qquad (18)$$

If it is assumed that

- The media is homogeneous and isotropic;
- Flow is one-dimensional with no lateral dispersion;
- The fluid properties are constant with time;
- The media porosity or void ratio is constant with time (no settlement);
- The fluid density is constant through the media profile;
- The fluid in the media pores does not freeze;
- The air phase remains at atmospheric pressure; and
- There are no thermoosmotic or osmotic gradients affecting flow,

then the general equation for partially saturated flow becomes (Richards, 1931):

$$\frac{\delta\theta}{\delta t} = \frac{\delta}{\delta z} \left(k(\theta) \frac{\delta\theta}{\delta z} \right) \qquad (19)$$

Since the volumetric water content, and hence the partially saturated hydraulic conductivity are a function of the depth, the general equation may be expanded to the Richard's Equation:

$$\frac{\delta\theta}{\delta\psi} \cdot \frac{\delta\psi}{\delta t} = \frac{\delta}{\delta z}\left(k(\theta)\,\frac{\delta\psi}{\delta z}\right) - \frac{\delta k(\theta)}{\delta z} \qquad (20)$$

The specific moisture capacity, $C(\psi)$, is equal to the slope of the media moisture/matric potential curve, or:

$$C(\psi) = \frac{\delta\theta}{\delta\psi} \qquad (21)$$

Therefore, the Richard's Equation may be modified as follows:

$$C(\psi)\,\frac{\delta\psi}{\delta t} = \frac{\delta}{\delta z}\left(k(\theta)\,\frac{\delta\psi}{\delta z}\right) - \frac{\delta k(\theta)}{\delta z} \qquad (22)$$

The first term in the equation for partially saturated flow through a porous media describes the distribution of water content as a function of time or change in media storage. The second term describes the flux due to hydraulic gradients and the third term describes the flux due to gravitational potential.

Under saturated conditions, the total head is equal to the pressure head and the hydraulic conductivity is a constant for the porous media and fluid. The unsaturated flow conditions, Equation (22) reduces to:

$$\frac{\delta\theta}{\delta t} = k\,\frac{\delta\phi}{\delta_z} = v_z \qquad (23)$$

which is the Darcy Equation for saturated flow.

Properties of a Homogeneous, Isotropic, Partially Saturated, Porous Media

The hydraulic conductivity and specific moisture capacity of homogeneous, isotropic, porous media vary as a function of the volumetric water content, θ, and the pressure head of matrix potential, ψ. Due to the nonlinearity of these functions, a number of numerical approximations have been developed based on curve fitting techniques. These relationships tend then to be historetic, in that the relationship between volumetric water content and pressure head is dependent on whether the porous media is draining or wetting. The equation of volumetric water content for drainage conditions developed by Brooks and Coray (1964) was:

$$\theta(\psi) = \begin{cases} n & \psi>\psi_b \\ \theta_r + (n-\theta_r)\left(\dfrac{\psi}{\psi_b}\right)^{-1} & \psi\le\psi_b \end{cases} \qquad (24)$$

where;

θ_r = the residual water content,

ψ_b = the air entry pressure (saturation),

n = the soil porosity, and

λ = the pore size distribution coefficient.

The specific moisture capacity, $C(\psi) = \delta\theta/\delta\psi$, is a discontinuous function:

$$C(\psi) = \begin{cases} 0 & \psi\le\psi_b \\ -\lambda(n-\theta_r)\ \dfrac{1}{\psi}\left(\dfrac{\psi}{\psi_b}\right)^{-\lambda} & \psi>\psi_b \end{cases} \qquad (25)$$

King (1965) developed a hyperbolic drainage model:

$$\theta(\psi) = (n\ \delta)\left[\frac{\cosh\left[\left(\dfrac{\psi}{\psi_o}\right)^{\beta} + \varepsilon\right] - \gamma}{\cosh\left[\left(\dfrac{\psi}{\psi_o}\right)^{\beta} + \varepsilon\right] + \gamma}\right] \qquad (26)$$

where: δ, ψ, β, ε, and γ are curve fitting parameters.

Rogowski (1971) used a logarithmic relationship for volumetric water content as a function of the pressure head:

$$\theta(\psi) = \begin{cases} n & \psi > n \\ n+\alpha\ \ln(\psi_b - \psi + 1) & \psi \le \psi_o \end{cases} \qquad (27)$$

where;

$$\alpha = \frac{(\theta_{15} - n)}{\ln\ (\psi_b - \psi_{15} + 1}\ , \qquad (28)$$

$\psi_{15} = 1.5 \times 10^4$ cm, and

θ_{15} = moisture content when $\psi = \psi_{15}$.

Clapp and Hornberger (1978) developed a mathematical model for the wetting of soil using a power law and a portion of a parabola. For a degree of saturation < 92% the pressure head is given by:

$$\psi = \psi_s W^{-b} \qquad (W \leq 0.92) \tag{29}$$

where;

W = the soil wetness = $\frac{\theta}{\theta_s} = \frac{S}{100}$

θ_s = saturated volumetric water content,

ψ_s = saturated pressure head, and

b = an empirical constant.

When degree of saturation is between 92% and 100% the pressure head is given by:

$$\psi = -m\,(W-n)(W-1) \qquad (0.92 < W \leq 1) \tag{30}$$

where;

$$m = \frac{\psi_i}{(1-W_i)^2} - \frac{\psi_i\, b}{W_i(1-W_i)} \tag{31}$$

$$n = 2W_i - \frac{\psi_i\, b}{m\, W_i} - 1, \quad \text{and} \tag{32}$$

ψ_i = the pressure head at W = 0.92.

Milly and Eagleson (1980) fit the nonlinear relationship between the volumetric water content and the log of the pressure, pF = log $(-\psi)$, with three line segments as shown in Figure 2. The equation given for the volumetric content is:

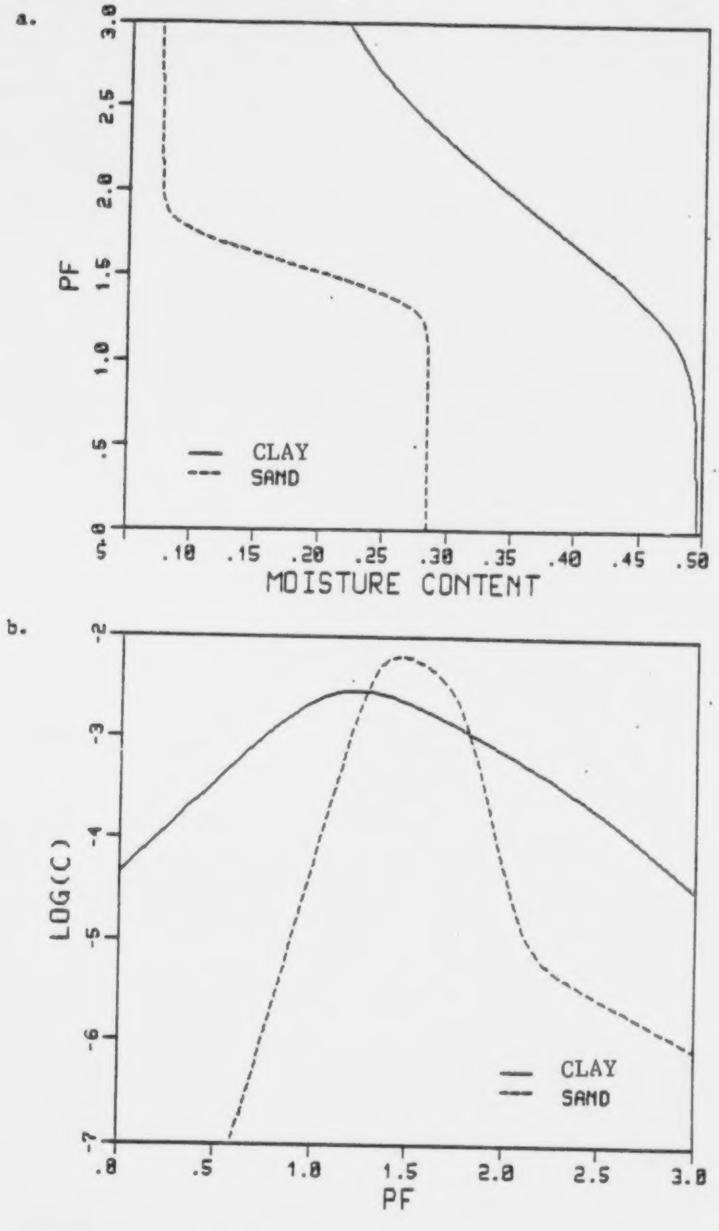


FIGURE 2. Sandy and clayey soil moisture characteristics computed from a continuous functional relationship, a. moisture content and b. log of specific moisture capacity (After Johnson et al., 1983)

$$\theta(pF) = \frac{1}{M} \ln\left\{ \exp\Big[M[a_1 - s_1(pF)]\Big] + \exp\Big[M[a_2 - s_2(pF)]\Big] \right\}$$

$$- \frac{1}{M'} \ln\left\{ \exp\Big[M'\ [a_2 - s_2(pF)]\Big] + \Big[\exp\ [M'\theta_u]\Big] \right\} + \theta_u \qquad (33)$$

where; M and M' are constants which control the curvature of the joining segments.

For the linear portion of the capillary range between pF_{min} and pF_o, Equation 33 reduces to:

$$\theta(\psi) = a_2 - s_2 \log(-\psi), \qquad (34)$$

and the specific moisture capacity is given by

$$C(\psi) = - \frac{s_2}{\ln(10)} \left(\frac{1}{\psi}\right) \qquad (35)$$

The specific moisture capacity for all values of ψ is continuous and is given by:

$$C(\psi) = \frac{1}{M}\left[p_1(-\psi)^{q_1} + p_2(-\psi)^{q_2}\right]^{-1} \left[-p_1 q_1(-\psi)^{q_1-1} \quad -p_2 q_2(-\psi)^{q_2-1}\right]$$

$$- \frac{1}{M'}\left[p_3(-\psi)^{q_3} + p_4\right]^{-1} \left[-(p_3)^{q_3}\ (-\psi)^{q_3-1}\right] \qquad (36)$$

where;

$$p_1 = e^{Ma_1}, \qquad (37)$$

$$p_2 = e^{Ma_2}, \qquad (38)$$

$$p_3 = e^{M'a_2}, \qquad (39)$$

$$p_4 = e^{M'u}, \qquad (40)$$

$$q_2 = -Ms_2 \log(e), \text{ and} \qquad (42)$$

$$q_3 = -M's_2 \log(e). \qquad (43)$$

These relationships for typical sand and clay soils are shown in Figure 3.

Partially Saturated Hydraulic Conductivity

The relationship between $k(\psi)$ and θ for a homogeneous, isotropic porous media is nonlinear, as shown in Figure 4.

Several numerical models have been developed for partially saturated hydraulic conductivity:

- Logarithmic Model (Gardner, 1952):

$$k(\psi) = \frac{a}{(b+(-\psi)^m)} \qquad (44)$$

where; a,b and m are curve fitting constants.

- Exponential Model (Gardner, 1958):

$$k(\psi) = k_s \exp(-a\psi) \qquad (45)$$

where; k_s = the saturated hydraulic conductivity.

- Power Law Model (Brooks and Corey, 1964):

$$k(\psi) = k_s \left(\frac{\psi}{\psi_b}\right)^{-\lambda} \qquad (46)$$

- Power Law Model (Campbell, 1974):

$$k = W^{2b-3} \qquad (47)$$

where;

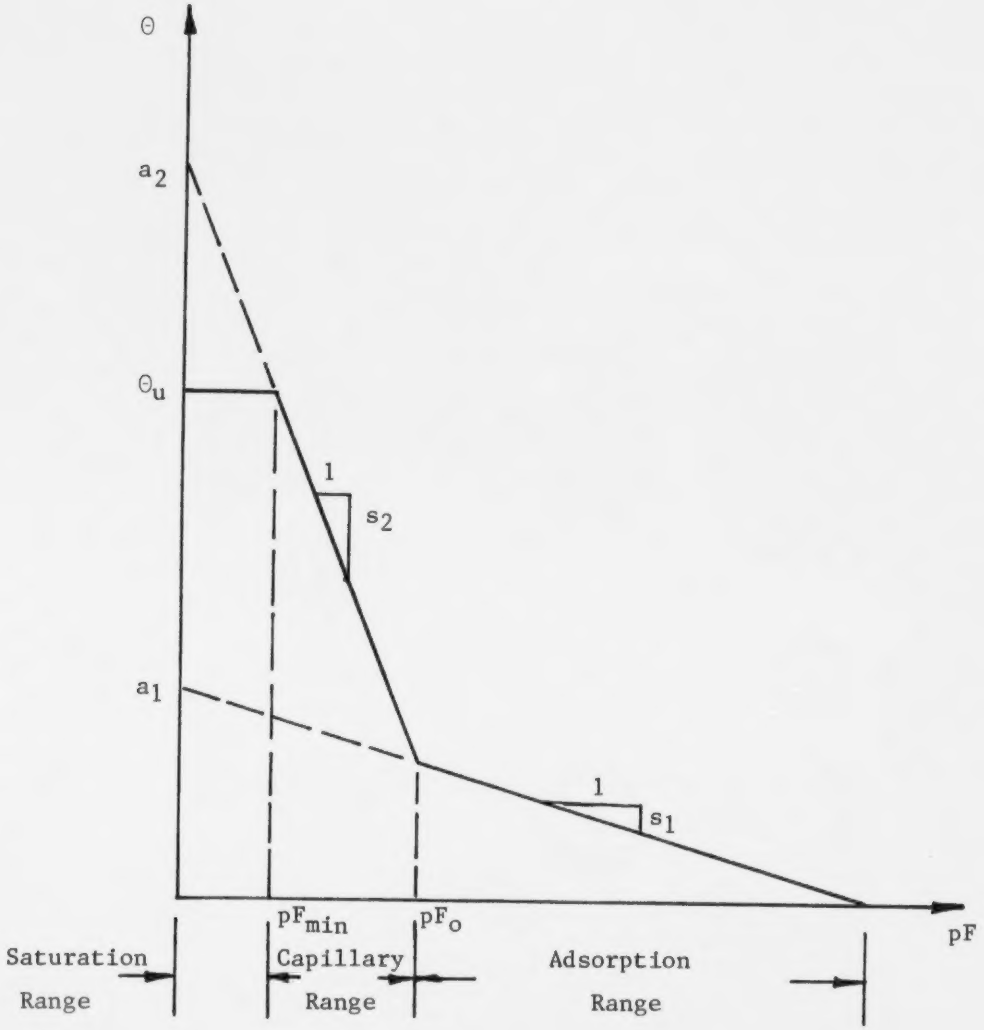

Θ
a_2
Θ_u
a_1
1
s_2
1
s_1
pF_{min}
pF_o
pF
Saturation Range
Capillary Range
Adsorption Range

Figure 3. Specific Moisture Content of Soils (After Johnson et al., 1983)

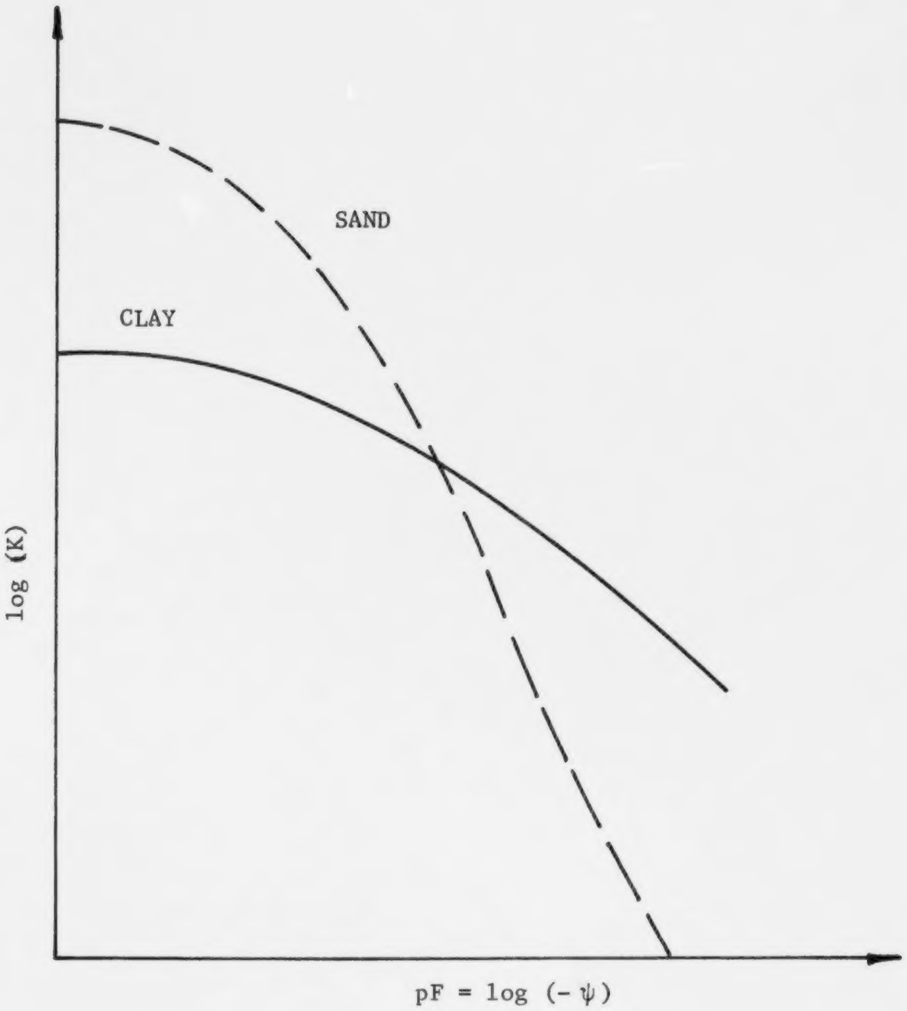


Figure 4. Schematic of unsaturated hydraulic conductivity for a sand and clay soil (After Johnson et al., 1983)

$$k = \frac{K}{K_s} \text{ , and} \tag{48}$$

b = curve fitting parameter related to the soil structure and gradation.

- Power Law Model (Maulem, 1978):

$$k(\psi) = k_s(S_e)^{0.015w\ +3.0} \tag{49}$$

where;

$$S_e = \text{the effective saturation} = \frac{\theta-\theta_r}{n-\theta_r} \tag{50}$$

$$w = \int_{\psi=+\infty}^{\psi=0} \gamma_w \psi \, d\theta \quad \text{and} \tag{51}$$

γ_w = the unit weight of water.

The equation for w represents work required to drain one pore volume of a homogeneous, saturated porous media. Maulem (1978) showed that this model was a significant improvement over previous models for the prediction of unsaturated hydraulic conductivity as a function of the volumetric water content.

Applications to Waste Materials

Numerous numerical models have been developed to model one-dimensional moisture migration in partially saturated soils. These models typically utilize finite element or finite difference methods for solution. Johnson, *et al*. (1983) performed a detailed review and comparison of the following models through a layered soil profile:

- UNSAT2 (Neuman, 1973; Neuman, *et al*., 1975)
- Lappela (1981)
- Reeves and Duguid (1975)
- FEMWATER (Yeh and Ward, 1980)
- TRVST (Nardsimhan, 1975)
- FLUMP (Neuman and Nardsimhan, 1977)
- TRANSAT (Pickens, Gillham, and Camran, 1979)

These simulations are useful for comparison of the various models. However, there are several significant differences between models used for evaluation of flow through partially saturated soils and models which will ultimately be developed for flow through waste in landfills.

The application of the partially saturated flow equation and specific moisture capacity and hydraulic conductivity relations to waste materials typically involves a great deal of guess work. Until field capacity is reached, the flow regime in a partially saturated waste cell is three-dimensional. The assumption of one-dimensional flow is a gross oversimplification of the problem. Moreover, experience indicates that leachate is generated before field capacity is reached. This leachate is generated primarily by short-circuiting through the landfill, although gravity flow of liquid waste materials may also have an impact on initial leachate quantity and characteristics. Therefore, the numerical model used to model the generation of leachate should incorporate short-circuiting.

As a further complication, waste materials are utilized or converted during the stabilization process. The surface area of the waste materials tend to increase with time and fractions of the waste may be consumed. This results in changes in the porosity, structure and even the composition of the waste. The existing partially saturated flow models do not account for changes in waste properties, whether incurred due to biological or physical (compression) mediation. These changes may significantly affect the partially saturated hydraulic conductivity of the waste. Unfortunately, little data is available to define these changes in either temporal or spacial dimensions.

BIOLOGICAL MODULE

The biological module of the GTLEACH-I model is a mechanistic three-step model which extends the two-step model proposed by Straub and Lynch (1982). It incorporates components from several other models and analyses within the perspective of the sequences of events constituting the phases of landfill stabilization. Consequently, a landfill may be considered to be a fixed-film microbial treatment process operating in a batch-wise configuration with continuous dilution and washout. The system is a batch process since the substrate, the degradable fraction of refuse, is depleted over time. Usually, both reactants and products of landfill stabilization processes are continuously or semi-continuously diluted and removed from the landfill in leachate and landfill gas. The microorganisms, as catalysts in the reaction, occur largely in films on refuse surfaces and also as aggregates in the interstices of the waste mass. The concentration of suspended microorganisms in the leachate is expected to be minimal due to the filtering action of the refuse, although experimental data confirming this supposition are currently lacking. Based on landfill stabilization trends observed by Pohland, et al. (1985), it is evident that the appearance and eventual disappearance of volatile acids in leachate are primary indicators of processes responsible for concurrent changes in COD, pH, and oxidation-reduction potential within the landfill. Thus, a model which is capable of simulating the changes in volatile acid concentrations in leachate over time could ultimately form the foundation for simulation of other indicator parameters such as pH and ORP along with corresponding changes in the mobility of hazardous constituents disposed in a landfill.

The three-step processes of hydrolysis/solubilization, acid formation and methane fermentation described previously were logical choices on which to base the development of GTLEACH-I. Since it has been applied successfully in the design and operation of other anaerobic processes (Grady and Lim, 1980; Henze and Harremoes, 1983; Mosey, 1983; Lindgren, 1983), it was reasonable to assume that the three-step process would also be successful in modeling anaerobic stabilization in landfill systems. Futhermore, such a mechanistic model could predict the concentration of volatile acid intermediates as a function of time as well as the rate of conversion of leachate COD to methane.

Hydrolysis/solubilization is an important process in the degradation of organic matter, and it is considered by Eastman and Ferguson (1981) to be the rate-limiting step in the acid phase of anaerobic digestion. The rate of hydrolysis is affected by many factors including pH, temperature, microbial biomass and associated substrate, the remaining concentration of particulate substrate, and hydrolysis product concentration. It is evident that any one or all of these factors may change with time in a batch-wise operation such as a landfill.

Eastman and Ferguson (1981) postulated that, at constant temperature and pH, hydrolysis is approximately first-order with respect to particulate or solid organic substrate concentration. This approach was used to model hydrolysis/solubilization in GTLEACH-I as expressed by the equation.

$$\frac{dM}{dt} = -K_H M \qquad (52)$$

where;

M = solid substrate concentration expressed as if it were suspended in landfill moisture, and

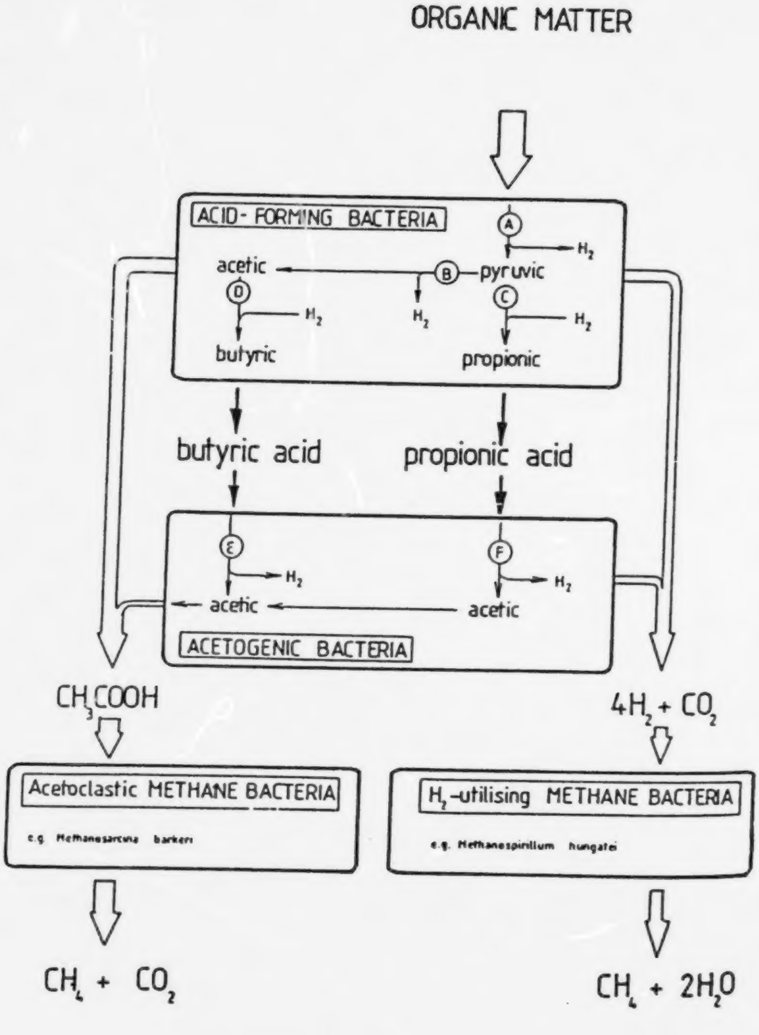
K_H = hydrolysis/solubilization rate constant.

While the approximation of first-order kinetics for hydrolysis is reasonable when considering the acid formation and methane fermentation phases of landfill stabilization separately, difficulty is expected in modeling the transition between the two phases when pH and other leachate characteristics change dramatically. Additionally, as relatively easily hydrolyzed substrates are exhausted and more refractory substrates begin to be utilized, a corresponding change in the rate of hydrolysis should be manifest.

Mosey (1983) summarized the mechanisms involved in conversion of glucose to carbon dioxide and methane as presented in Figure 5. Acid forming bacteria, the fastest growing of the four depicted groups of bacteria, ferment glucose to produce a mixture of reduced end products, e.g., acetic, propionic and butyric acids. Other volatile acids are produced as well in

FIGURE 5

MICROBIAL ECOLOGY OF THE ANAEROBIC DIGESTION PROCESS
(Mosey, 1983)


ORGANIC MATTER
ACID-FORMING BACTERIA
A
H_2
acetic
B
pyruvic
D
H_2
H_2
C
H_2
butyric
propionic
butyric acid
propionic acid
E
H_2
F
H_2
acetic
acetic
ACETOGENIC BACTERIA
CH_3COOH
$4H_2 + CO_2$
Acetoclastic METHANE BACTERIA
e.g. Methanosarcina barkeri
H_2-utilising METHANE BACTERIA
e.g. Methanospirillum hungatei
$CH_4 + CO_2$
$CH_4 + 2H_2O$

the fermentation of soluble organic matter other than glucose. Acetogenic bacteria convert long-chain volatile fatty acids into acetic acid. The existence of these acetogenic bacteria has not yet been proven except by deduction from the inability of any known methanogenic bacteria to metabolize propionate and butyrate. The growth rates of acetogenic bacteria are relatively low in comparison with acid-forming bacteria and can be easily product-inhibited by the accumulation of dissolved hydrogen gas in the growth media. An analogous though somewhat more complicated pathway exists for mixtures of complex soluble substrate expected from the hydrolysis of the degradable fraction of waste disposed in a landfill.

Production of volatile acids was represented in GTLEACH-I by a single Monod equation for a generalized group of acid-forming bacteria rather than a more complicated set of equations for multiple populations of bacteria.

The overall growth rate for this group of bacteria was described by the equation:

$$\frac{dX_A}{dt} = \frac{Y_A \, Q_A \, S_H \, X_A}{K_A + S_H} - K_{DA} \, X_A \tag{53}$$

where;

X_A = acidogen concentration (mg/L),

S_H = hydrolysis/solubilization product concentration (mg/L),

Q_A = maximum substrate utilization rate for acidogens (days^{-1})

K_A = half-velocity constant for acidogensis (mg/L),

Y_A = yield coefficient for acidogens (mass acidogens produced per mass substrate removed), and

K_{DA} = decay rate for acidogens.

The major assumption underlying the selection of a single equation to model behavior of a group of microorganism is that the relative activities of the various populations of microorganism are constant with respect to each other so that the net reaction follows Monod kinetics. While the validity of this assumption may be questioned when applied to landfills, the single-equation Monod model of acidogenesis was selected to allow application of the model to the database currently available and to minimize the number of input parameters to the model.

Methane is produced from acetic acid by acetoclastic bacteria and from hydrogen and carbon dioxide by hydrogenophilic bacteria. Bacteria producing methane from hydrogen and carbon dioxide are faster growing than those metabolizing acetic acid, but because of their mutualism, the rate of methane production is limited by the slower growing acetophilic bacteria (Henze and

Harremoes, 1983). For this reason, a single Monod equation for methanogenic microbial activity was employed in GTLEACH-I rather than two separate equations, also serving to minimize the number of input parameters to the model. The expression for the overall growth rate of the two groups of methanogenic bacteria was employed in GTLEACH-I was:

$$\frac{dX_M}{dt} = \frac{Y_M Q_M S_H X_M}{K_M + S_A} - K_{DM} X_M \qquad (54)$$

where;

S_A = volatile acid concentration (mg/L),

X_M = methanogen concentration (mg/L),

Q_M = maximum substrate utilization rate for acidogens (days^{-1}),

K_M = half-velocity constant for methanegens (mg/L),

Y_M = yield coefficient for methanogens (mass methanogens produced per mass substrate removed), and

K_{DM} = decay rate for acidogens.

One other aspect of methanogenesis to be considered is that methanogen growth and concurrent methane production can be suppressed at pH levels below the range of 6.8 to 7.5. This explains the virtual absence of methane production observed during the acid phase of landfill stabilization when the pH of the landfill moisture can be well below 6. That methanogenesis is established at all is attributed to the heterogenous nature of landfills which allows methanogens to begin to grow in small pockets of the landfill not completely saturated with volatile acids. Methanogen growth gradually spreads out from these pockets until a methanogen population is established sufficient to reduce the concentrations of volatile acids in the landfill moisture and increase the pH, thereby allowing unsuppressed methanogen growth throughout.

As a first attempt at modeling landfill processes, it was decided to provide for suppression of methanogenesis during the acid phase by simply delaying initiation of methanogenesis for a specified period of time in GTLEACH-I. Further work in the area of landfill modeling will require development of the pH-inhibition term in Equation 54 for methanogenesis which will allow for methanogen growth at a suppressed rate during periods of low pH levels in leachate.

Diffusional resistance to reactions in a dense matrix such as a microbial film or aggregate may have significant influence on reaction kinetics by reducing the efficiency of the microbial mass (Henze and Herremoes, 1983). Due to difficulties encountered in evaluating the total microbial mass, the thickness of biofilms or aggregates, and due to heterogeneities in landfill environments, diffusional resistances were not explicitly treated in GTLEACH-I. Instead, they were subsumed in the fitting of the Monod rate constants to experimental data. Deviations of the fitted kinetic parameters from literature values were expected to reflect not only diffusional resistances, but also the effects of environmental heterogeneity and differences in the type of solid substrate degraded.

On the basis of the preceding discussion, the following assumptions were made in constructing the biological module of the GTLEACH-I model:

- Solubilization/hydrolysis of solid substrate to soluble intermediates were considered a first-order reaction with respect to mass of solid substrate concentration or surface area.

- Production of volatile acids from soluble intermediates by a group of acidogens was considered a function of hydrolysis product concentration according to Monod kinetics as well as a first-order function of acidogen concentration.

- Production of methane from volatile acids by methanegens was considered a function of volatile acid concentration according to Monod kinetics and also a first-order function with respect to methanogen concentration.

- The mixing regime of a single lift in a landfill was modeled as a number of equal-sized, completely-mixed reactors in series.

- By the time leachate generation begins, field capacity had been attained and remained constant thereafter.

- The flows of leachate into a landfill lift, through each reactor within the lift, and out of the landfill lift were equal at any instant in time so that there was no accumulation of moisture in any portion of the landfill above field capacity. The flow values provided in GTLEACH-I were constant, average values of daily flow. (An assessment of the impacts of these flow assumptions is contained herein. Since GTLEACH-I has not yet been combined with a flow rate were necessary. When this work is integrated with a companion flow model, flow and field capacity should be allowed to vary from day-to-day in accordance with rates of moisture applications and drainage.)

- Solid substrate was also assumed to be fixed within the landfill.

- Hydrolysis products and volatile acids existed only as part of the landfill leachate, i.e., they did not sorb or precipitate, and, therefore, could be washed completely out of the landfill given sufficient flow.

- Since inhibition or delay in the establishment of methanogenesis by environmental conditions is a commonly observed phenomenon in anaerobic processes, a simplistic approach for modeling the phenomenon was taken in GTLEACH-I by providing for a lag time of specified length before establishment of methanogenesis.

Based on these assumptions, the GTLEACH-I model was constructed of five differential equations solved simultaneously for each of a series of continuously-stirred tank reactors. The total volume of moisture or field capacity contained in the landfill lift or experimental landfill cell was divided evenly among the tanks-in-series. The rate of change of solid substrate mass, expressed as concentration in the volume of moisture contained within a reactor, was a first-order decay as given by Equation 55:

$$\frac{dM}{dt} = -K_H M \tag{55}$$

The rate of change of acidogen and methanogen masses, also expressed as concentration in the volume of moisture contained within a reactor, followed Monod kinetics as given by Equations 56 and 57:

$$\frac{dX_A}{dt} = \frac{Y_A Q_A S_H X_A}{K_A + S_H} - K_{DA} X_A \tag{56}$$

$$\frac{dX_M}{dt} = \frac{Y_M Q_M S_H X_M}{K_M + S_A} - K_{DM} X_M \tag{57}$$

The rate of change of hydrolysis product concentration in a continuously-stirred tank reactor was a function of four processes: the rate of production by hydrolysis, the rate of removal by acidogenesis, the rate of flow of hydrolysis products dissolved in landfill moisture passing into the reactor from the preceding reactor, and the rate of flow of hydrolysis products carried out of the reactor and into the next. The differential equation describing these simultaneous processes is given as:

$$\frac{dS_H}{dt} = K_H M - \frac{Q_A S_H X_A}{K_A + S_H} + \frac{F}{V}(S_{HO} - S_H) \tag{58}$$

Likewise, the rate of change of volatile acid concentration in a continuously-stirred tank reactor was also a function of four processes: the rate of production of volatile acids by acidogenesis, the rate of removal by methanogenesis, the rate of flow of volatile acids dissolved in moisture passing into the reactor from the preceding reactor, and the rate of flow of volatile acids carried out of the reactor and into the next. The differential equation describing these simultaneous processes is given as:

$$\frac{dS_A}{dt} = \frac{Q_A S_H X_A}{K_A + S_H} - \frac{Q_M S_A X_M}{K_M + S_A} + \frac{F}{V}(S_{SAO} - S_{SA}) \qquad (59)$$

In Equations 55, 56, 57, 58, and 59:

M = solid substrate concentration as if it were suspended solids in landfill moisture (mg/L),

K_H = hydrolysis/solubilization rate constant (days^{-1}),

X_A = acidogen concentration as if suspended in landfill moisture (mg/L),

S_H = hydrolysis/solubilization product concentration in landfill moisture (mg/L),

Q_A = maximum substrate utilization rate for acidogens (days^{-1}),

K_A = half-velocity constant for acidogenesis (mg/L),

Y_A = yield coefficient for acidogens (mass acidogens produced per mass substrate removed),

K_{DA} = decay rate for acidogens,

S_A = volatile acid concentration in landfill moisture (mg/L),

X_M = methanogen concentration as if suspended in landfill moisture (mg/L),

Q_M = maximum substrate utilization rate for methanogens (days^{-1}),

K_M = half-velocity constant for methanogens (mg/L),

Y_M = yield coefficient for methanogens (days^{-1}),

K_{DM} = decay rate for methanogens (days^{-1}),

F = moisture flow rate through reactor (L/day),

V = volume of reactor (L),

S_{HO} = influent hydrolysis/solubilization product concentration (mg/L), and

S_{AO} = influent volatile acid concentration (mg/L).

These five equations were solved simultaneously to give concentrations of solid substrate, acidogens, methanogens, hydrolysis products, and volatile acids as a function of time in the moisture contained in each of the reactors in series. The composition of the moisture in the final reactor in the series was that of the leachate emanating from a single experimental cell or landfill lift. The composition of moisture entering the first reactor in a series was that of pure water for a landfill cell operated in single-pass mode. For an experimental landfill cell operated in recycle mode, the composition of moisture entering the first reactor in series was equivalent to the leachate composition exiting the final reactor in the series.

the series.

ORGANIZATION OF GTLEACH-I

GTLEACH-I is currently composed of one main program, PROGRAM GTLEACH-I and five subroutines: INPUT, OUTPUT, FLOW, FCN, and FCNJ. A copy of the FORTRAN source code for GTLEACH-I is provided in Appendix B.

GTLEACH-I is linked to a single precision IMSL Math/PC library (1984) which contains DGEAR, a differential equation solving subroutine for initial value problems composed of systems of differential equations which must be solved simultaneously. Subroutine FCN is a subroutine called by DGEAR containing the system of differential equations to be solved. When DGEAR is called, DGEAR solves the differential equations given in FCN from initial values taken from the current value of the matrix Y(N) and the independent variable X. In this application:

X = time (days),
Y(1) = solid substrate concentration (mg/L),
Y(2) = acidogen concentration (mg/L),
Y(3) = methanogen concentration (mg/L),
Y(4) = hydrolysis product concentration (mg/L), and
Y(5) = volatile acid concentration (mg/L).

DGEAR returns the solutions as new values in X and Y(N). Detailed specifications for FCN and FCNJ (a dummy subroutine in this application) as well as guidance for the use of DGEAR are given in the DGEAR section of the IMSL Library Users Manual [IMSL, 1984].

The underlying structure of PROGRAM GTLEACH-I is composed of a pair of nested Do-loops, (DO 175 I-1, NDAYS) and (DO 150 J=1, NCSTR). The outer loop, the I loop, calls for the solution of the differential equations in steps of days, and the inner loop, the J loop, for solution at each CSTR of the landfill. DGEAR actually employs time-steps many orders of magnitude smaller than one day in solving the equations, however, the composition of the flow between CSTR's is only adjusted at the end of each day as controlled by the I loop. This size time step is adequately small as long as the solution exhibits the desired degree of smoothness.

GTLEACH-I allows for two modes of landfill operation, single-pass and recycle, as determined by the state of the variable IRECYC in the set of statements following the beginning of the I loop. If IRECYC is equal to zero, then the concentration of hydrolysis products and volatile acids in the influent to the first CSTR (SHO and SAO found in subroutine FCN) are set equal to zero, thereby resulting in a single-pass system. If IRECYC equals 1, then the concentration of SHO and SAO found are the same as in the effluent from the last CSTR of the previous time steps (Y(4) and Y(5)).

Y(N) is a matrix containing the solutions to the N differential equations for the most recently solved CSTR at the most recent time step. The values in Y(N) are overwritten each time DGEAR is called. The matrix YY(J,K) stores the values of the solutions Y(N) for an entire time step such that J is the CSTR number and K is the number of the differential equation. Thus, before DGEAR is called, the initial values of solution matrix Y(N) are taken as the final values from the previous time-step for the same CSTR in the do-loop (DO 50 K=1,N). The next statement sets the value of X, the initial value of time for the calculation corresponding one day from XEND, the end of the current time-step.

Once DGEAR is called and the solutions returned in Y(N), the influent concentrations of SHO and SAO for the next CSTR are set to the effluent values of the current CSTR from the previous time step as stored in YY(J,4) and YY(J,5). The values of YY(J,K) are overwritten with the current solutions of Y(N) in the do-loop (DO 100 K=1,N). The variable INDEX is reset to 1 so that DGEAR may be provided new initial values at the next call.

Subroutine INPUT reads the input file PARAM.DAT which contains the input parameters and run-specific information. The input format statements are list-directed so that the input parameters may be entered in any format and need only be separated by commas. Line 1 of PARAM.DAT should contain the parameters: N, METH, MITER, INDEX, H, and TOL. These are parameters required by subroutine DGEAR and are defined in the IMSL Library User's Manual (IMSL, 1984). These parameters need only be altered if the GTLEACH-I code is modified. The one parameter of interest in the list is N, the number of differential equations to be solved by DGEAR.

The second line of PARAM.DAT contains the parameters: TVOL, XMASSO, SHO, SAO, XAO, and XMO. They are defined as follows:

TVOL = total volume of moisture contained in the landfill at field capacity,
XMASSO = total mass of food waste or degradable waste initially disposed in the landfill,
SHO = concentration of soluble substrate in leachate at time leachate first appears,
SAO = concentration of volatile acid in leachate at time leachate first appears,
XAO = concentration of acidogens in landfill moisture at time leachate first appears, and
XMO = concentration of methanogens in landfill moisture at time leachate first appears.

The third line of PARAM.DAT contains the parameters IOUT and LAG. The value of IOUT determines whether the output values of leachate strength are given in concentration (mg/L) or in total mass contained in the leachate for one day's flow. A value of 0 for IOUT selects concentration and a value of 1 selects total mass. LAG allows for inhibition of methanogenesis by delaying the time at which methanogen growth is initiated. LAG should be provided as the number of days after leachate generation begins at which methanogen growth begins.

Line 4 of PARAM.DAT contains the parameters XKA, XKDA, and YA which are the Monod parameters for acidogens.

XKA = half-velocity constant for acidogens (mg/L),
XKDA = decay rate for acidogens (days^{-1}), and
YA = yield coefficient for acidogens (mass biomass produced per mass substrate removed).

Line 5 of PARAM.DAT contains the same parameters for methanogens:

XKA = half-velocity constant for acidogens (mg/L),
XKDA = decay rate for acidogens (days^{-1}), and
YA = yield coefficient for acidogens (mass biomass produced per mass substrate removed).

Line 6 contains the values for XKH, QA and QM, defined as:

XKH = first-order rate constant for hydrolysis/solubilization (days^{-1}),
QA = maximum rate constant for acidogenesis (days^{-1}), and
QM = maximum rate constant for methanogenesis (days^{-1}).

Finally, Line 7 contains the run-specific values for NCSTR, NDAYS, and IRECYC which are:

NCSTR = the number of CSTR's in series,
NDAYS = the number of days after leachate first appears for which GTLEACH-I should be run, and
IRECYC = 0 for single-pass mode, 1 for recycle mode.

Subroutine OUTPUT is called once every five time steps (days) to write the solution to a file called RESULT.DAT. RESULT.DAT contains 7 columns of effluent leachate data, namely: time (days) since leachate generation began, the mass of degradable substrate remaining, the mass of acidogens, the mass of methanogens, the mass of hydrolysis products, the mass of volatile acids and the mass of methane produced.

SECTION 6

EVALUATION OF GTLEACH-I

The GTLEACH-I model was applied in the simulation of a set of recent landfill stabilization experiments conducted by Schaffer (1986) and Yari (1986) which are the latest in a series of simulated landfill experiments by Pohland and coworkers. The experiments were designed to capture the temporal processes occurring within a single lift or cell of a landfill.

Two modes of leachate management were employed, single-pass and recycle. In the single-pass experiment, moisture infiltrated the landfill, percolated through, and exited from the bottom of the landfill much as in a typical municipal solid waste landfill. Management of leachate in the recycle mode involved collecting leachate from the bottom of the landfill and reapplying it to the surface, allowing leachate to percolate through the landfill repeatedly. Either mode of operation can be simulated by GTLEACH-I.

The results of fitting GTLEACH-I to the Schaffer (1986) and Yari (1986) data are discussed subsequently. However, before proceeding to fitting GTLEACH-I to experimental data, an examination of the various parameters output by the model for a typical simulation serves to illustrate the capabilities of GTLEACH-I. The example selected is the simulation of the Schaffer (1986) and Yari (1986) data which, using only a single group of parameters, provides the best possible fit to both the single-pass and recycle data. The only difference between recycle and single-pass simulation is that flow into the recycle cell has the same composition as leachate exiting the cell whereas the composition of moisture flowing into the single-pass cell was that of pure water. This simulation employs the following input parameters:

Flow, F = 6 L/wk = 0.86 L/day,
Moisture content, TVOL = 71 L,
Number of CSTRs in series, NCSTR = 3,
Initial mass of solid substrate, XMASSO = 11 kg,
Initial concentration of hydrolysis products, SHO = 40,000 mg/L,
Initial concentration of volatile acids, SAO = 5,000 mg/L,
Initial concentration of acidogens, XAO = 100 mg/L, and
Initial concentration of methanegens, XMO = 10 mg/L.

Kinetic parameters for the various processes are:

Hydrolysis	Acidogenesis	Methanogenesis
K_H = .0001/day	Q_A = 3.2/day	Q_M = 1.9/day

K_A = 200 mg/L K_M = 500 mg/L

K_{DA} = 0.5/day K_{DM} = 0.02/day

Y_A = 1.0 Y_M = 0.02

Methanogen lag = 200 days

Basic criteria for selection of these parameters are discussed subsequently. The simulations presented here provide an understanding of the general characteristics of single-pass and recycle simulations, but as will be demonstrated in the next section, details of the simulations can be modified by varying the choice of input parameters.

Figures 6 through 12 illustrate graphically the variation in values of the output parameters with time for the single-pass simulation. Over a period of nearly 450 days the solid substrate, expressed as concentration in landfill moisture, decreased from an initial value of 155,000 mg/L to a value of 148,000 mg/L due to hydrolysis/solubilization. This represented a decrease in available solid substrate of approximately five percent as shown in Figure 6. It was difficult to assess whether or not this was a realistic utilization of solid substrate because of the difficulty in measuring solid substrate experimentally. In this work solid substrate was taken as the food fraction of municipal solid waste, estimated to be 20% by weight (Tchobanoglous *et al.*, 1977). It is unlikely that the entire food fraction of municipal solid waste was degradable by anaerobic processes, however, a better approximation was unavailable.

The mass of acidogens, expressed as concentration in landfill moisture, reached a maximum value of more than 5,000 mg/L during the first week of leachate generation (Figure 7). The concentration of acidogens quickly decreased over the next two weeks, stabilizing at a value of approximately 300 mg/L. For the remainder of the simulation, the acidogen concentration profile exhibited a slight but steady decrease. Comparison of Figure 8 with Figure 7 indicated that trends in hydrolysis product concentration decreased from an initial value of 40,000 mg/L to a value of 37 mg/L. This value was maintained for the remainder of the simulation, indicating that the rate of appearance of hydrolysis products by hydrolysis equalled the rate of removal by acidogenesis and washout.

Methanogen growth was initiated on Day 200 in accordance with the imposed lag. As illustrated in Figure 9, the initial concentration of 10 mg/L was sustained for the first 21 days then steadily decreased to approximately 2 mg/L by Day 446.

Volatile acid concentration in the leachate (Figure 10) quickly increased from the initial value of 5,000 mg/L to 17,500 mg/L within the first week of leachate generation. The value subsequently decreased entirely under the influence of washout until methanogenesis initiated on Day 200 and further increased the rate of removal of volatile acids. The concentration of volatile acids had decreased to 200 mg/L by Day 260, then gradually trended upward to a value of 300 mg/L by Day 446.

FIGURE 6

SOLID SUBSTRATE CONCENTRATION IN LANDFILL MOISTURE,
SINGLE-PASS SIMULATION

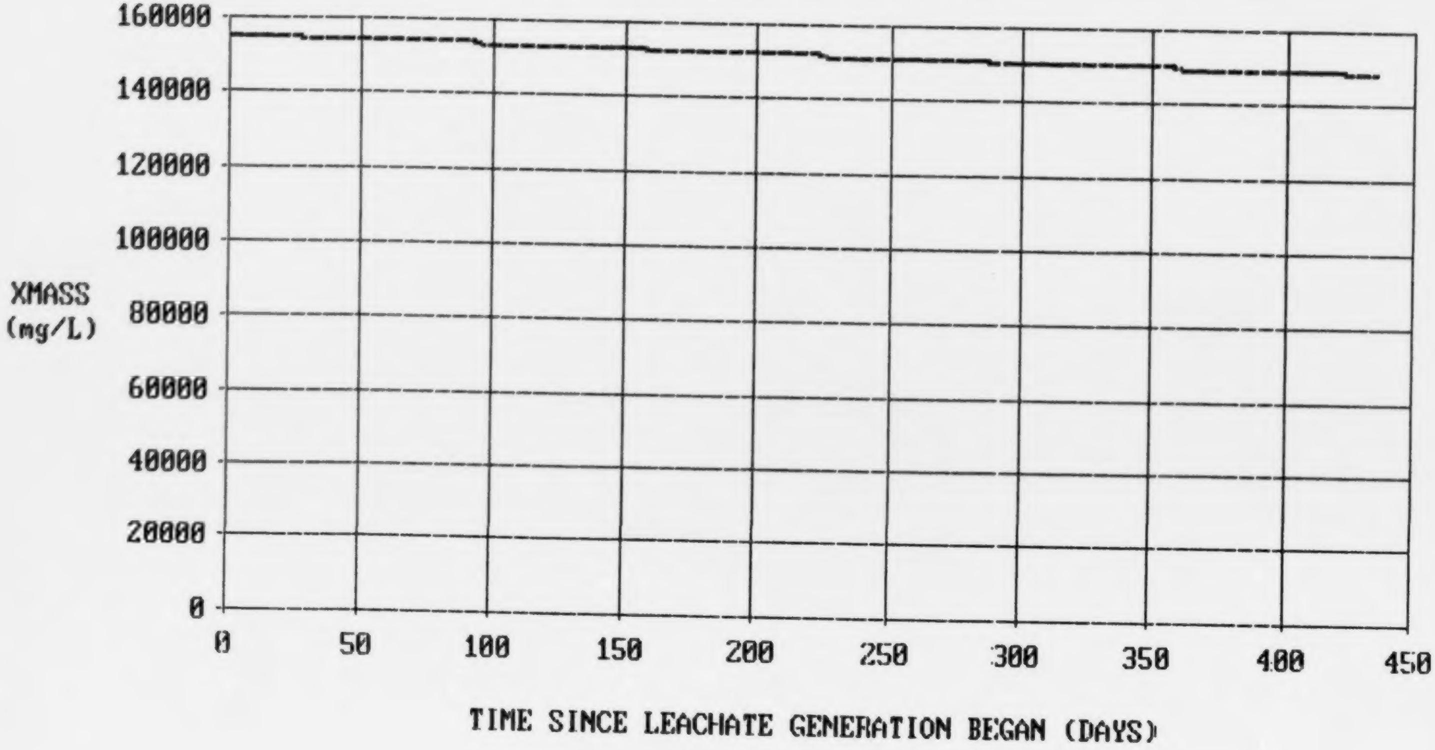


FIGURE 7

ACIDOGEN CONCENTRATION IN LANDFILL MOISTURE,
SINGLE-PASS SIMULATION

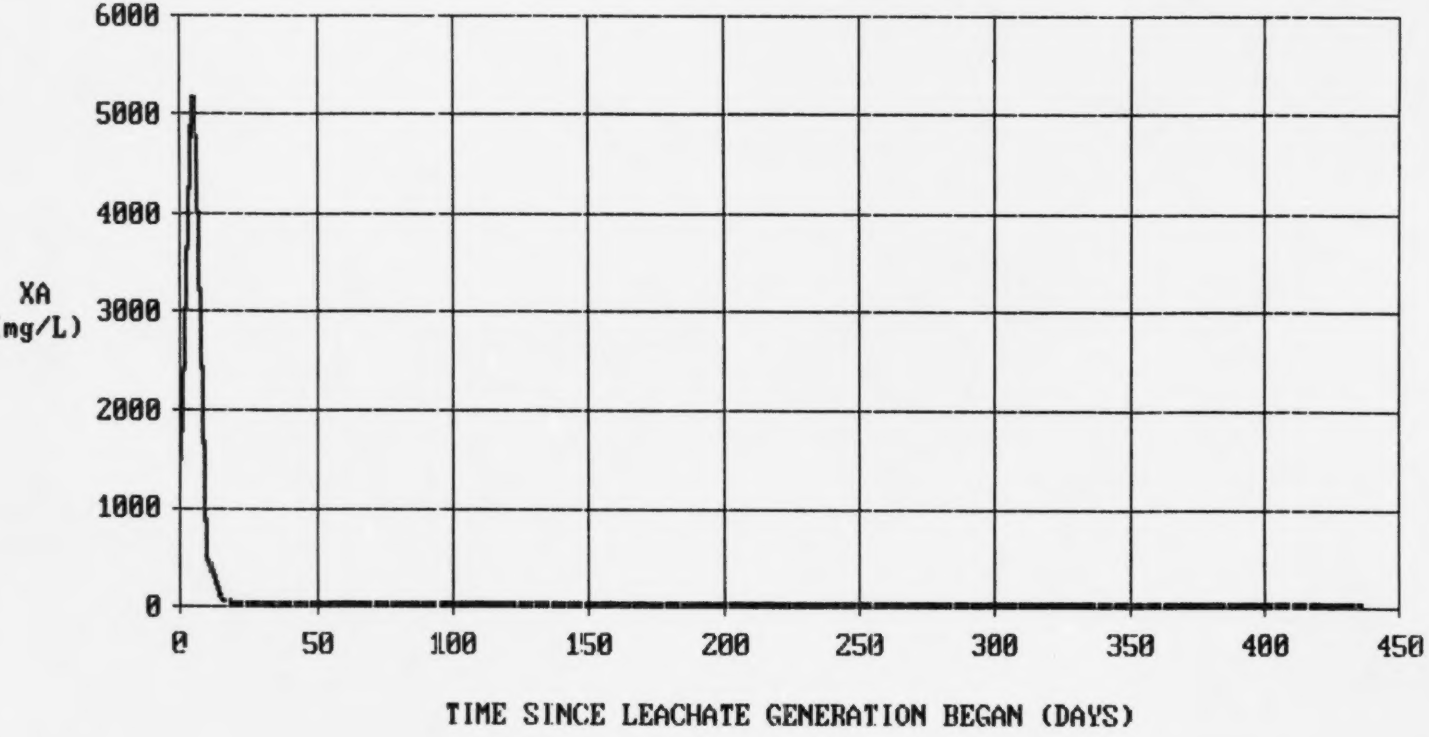


FIGURE 8

HYDROLYSIS PRODUCT CONCENTRATION IN LEACHATE,
SINGLE-PASS SIMULATION

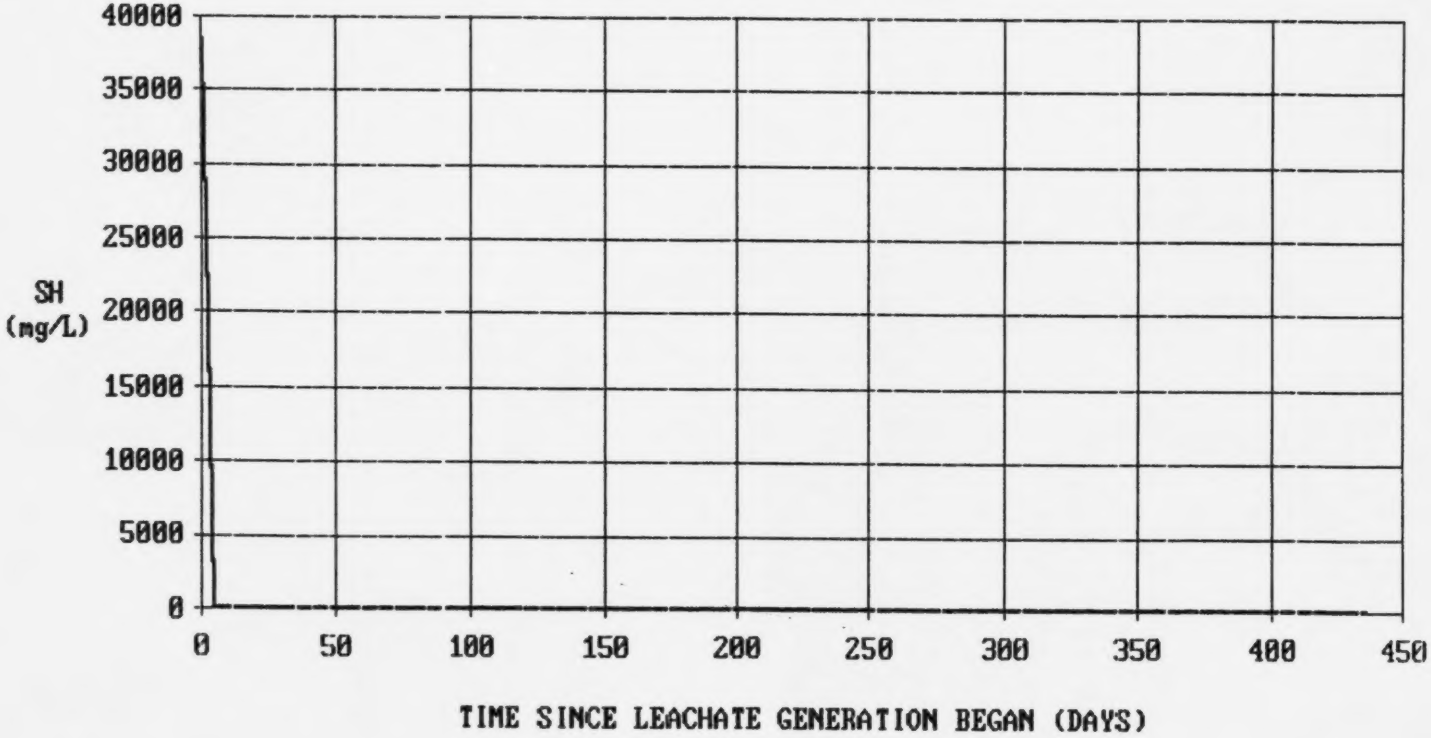


FIGURE 9

METHANOGEN CONCENTRATION IN LANDFILL MOISTURE,
SINGLE-PASS SIMULATION

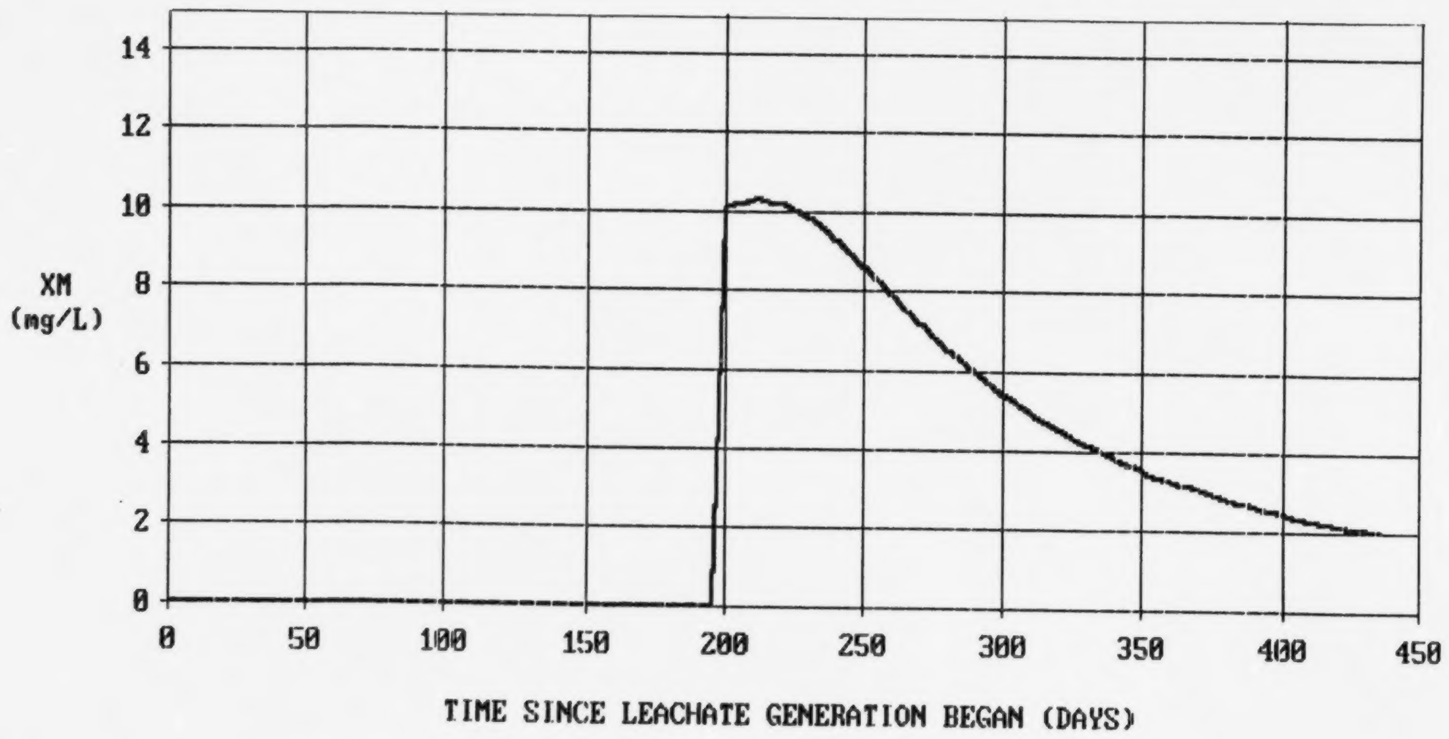


FIGURE 10

VOLATILE ACID CONCENTRATION IN LANDFILL LEACHATE,
SINGLE-PASS SIMULATION

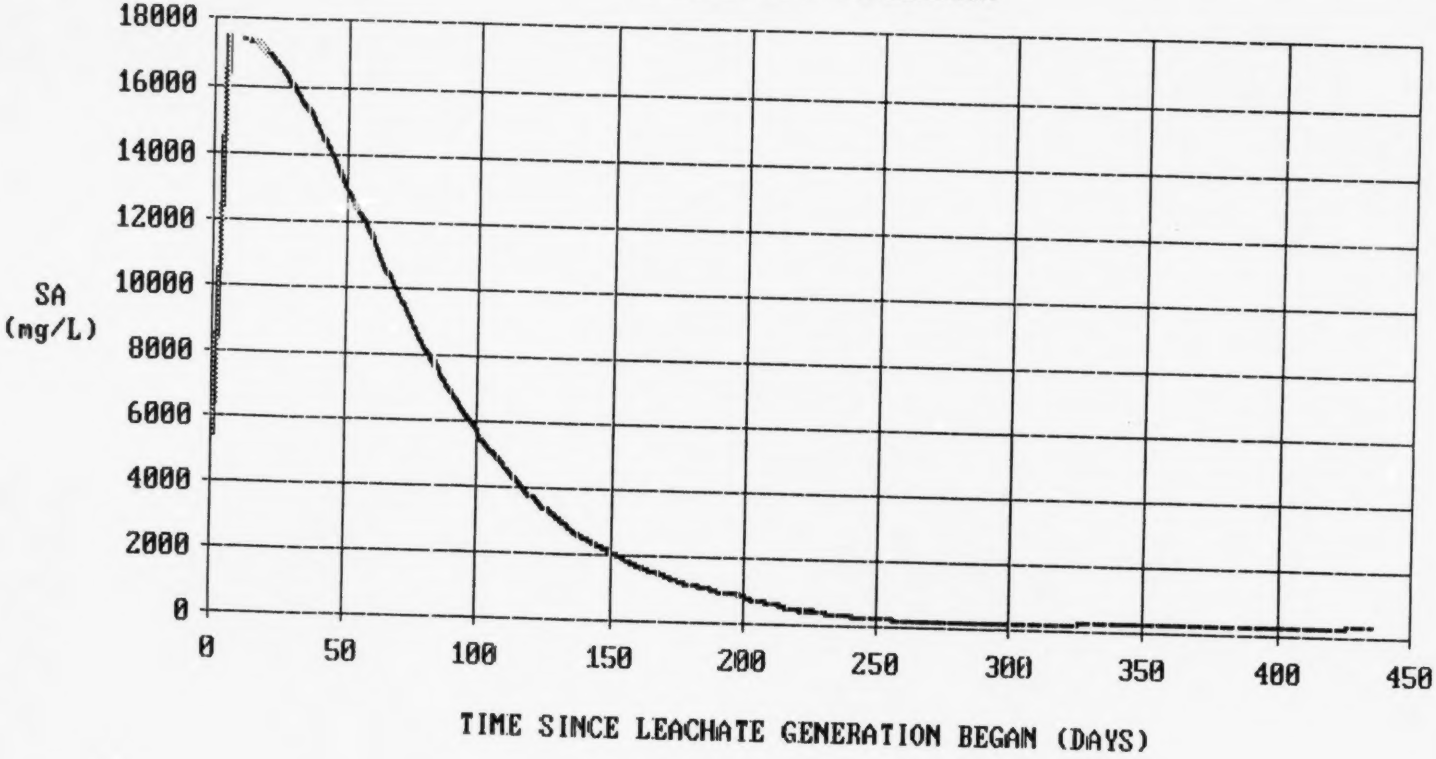


FIGURE 11

COD CONCENTRATION IN LANDFILL LEACHATE,
SINGLE-PASS SIMULATION

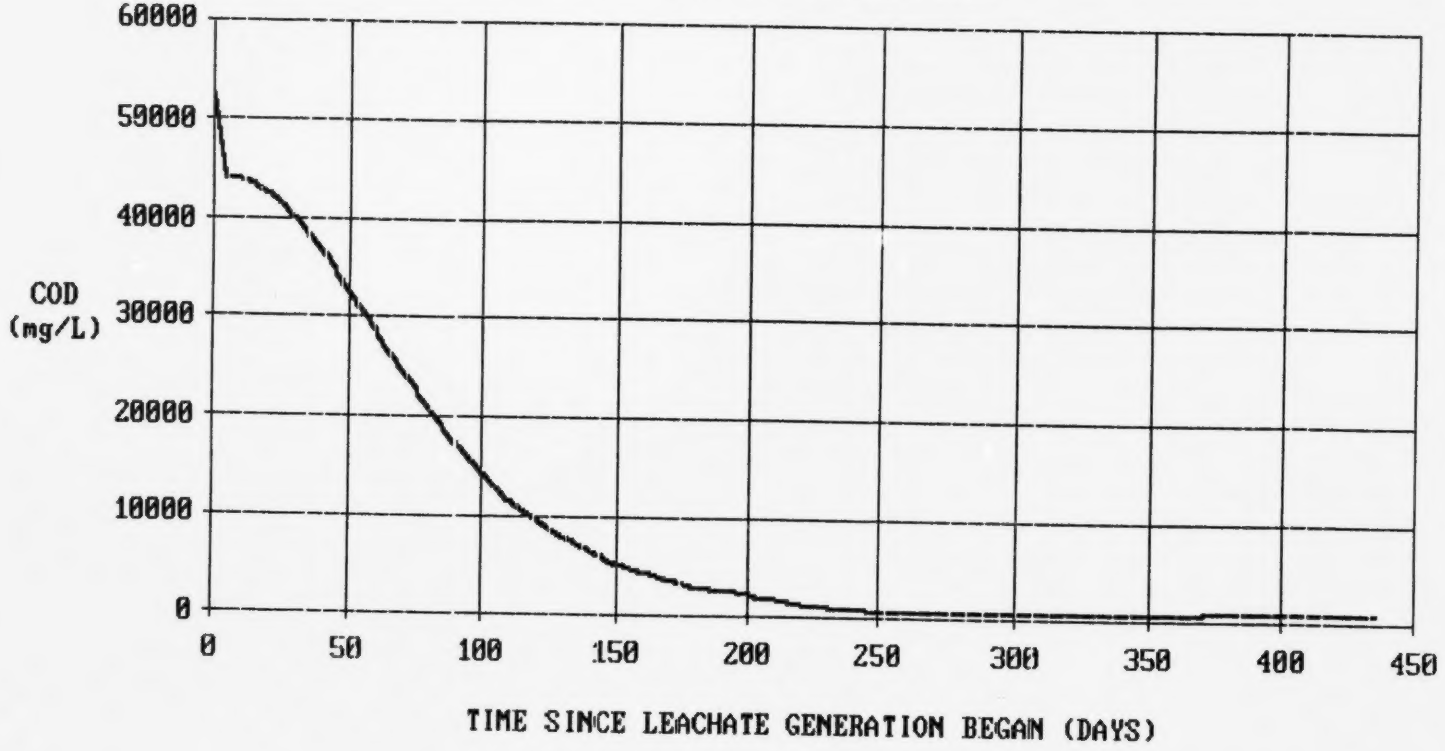
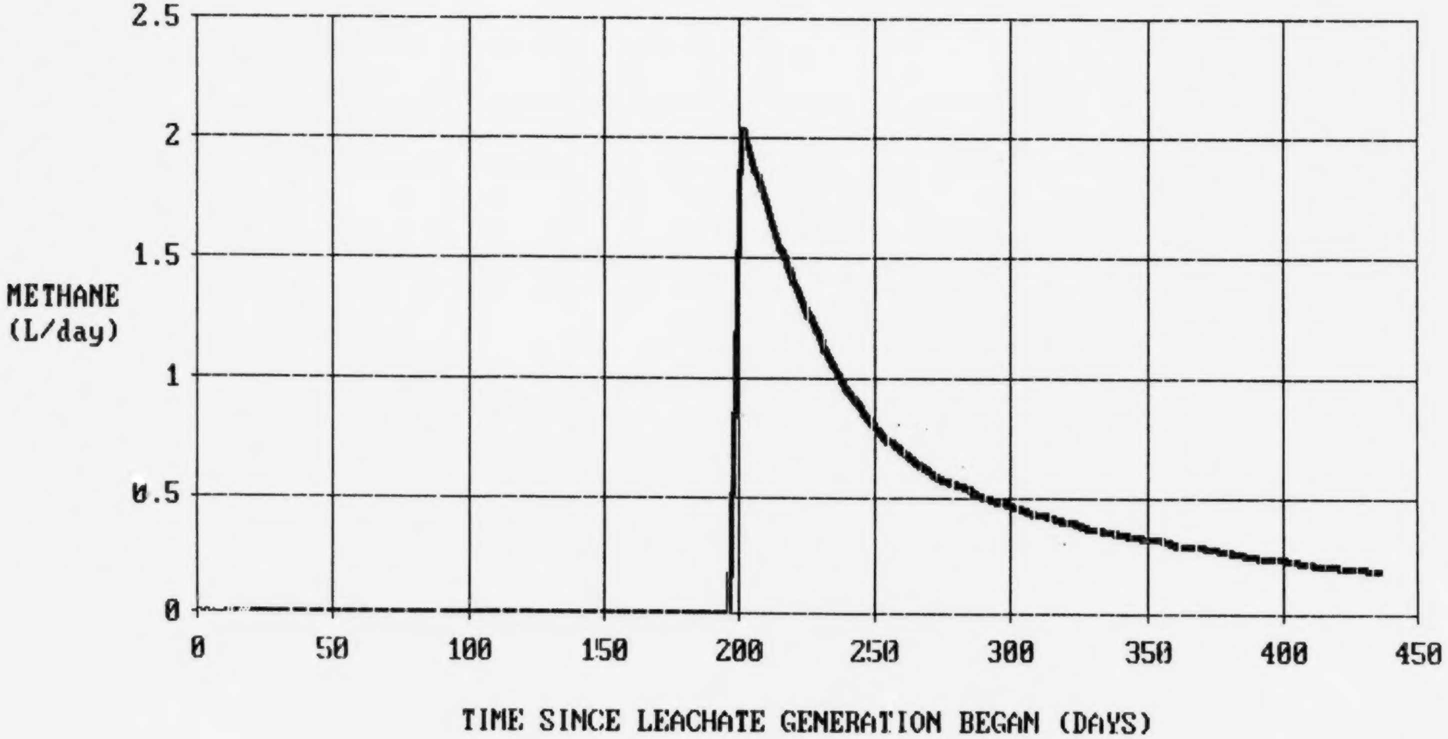


FIGURE 12

METHANE PRODUCTION,
SINGLE-PASS SIMULATION



The concentration profile of COD in leachate (Figure 11) was simply constructed as the sum of hydrolysis product concentration and volatile acid COD concentration. The shape of the COD concentration profile was similar to that of volatile acid concentration with the addition of an initial peak caused by the peak in hydrolysis product concentration.

Methane production (Figure 12) was initiated at a rate of 2 L/day, corresponding with the initial 10 mg/L concentration of methanogens. Methane production decreased as the supply of volatile acids was depleted, falling to a production level of half the initial value within 50 days.

Figures 13 through 19 illustrate the variation in values of the output parameters with time for the recycle simulation. Solid substrate removal in the recycle simulation exhibited the same five percent decrease over 450 days as the single-pass simulation (Figure 13). As discussed earlier, it was difficult to determine whether simulated levels of solid substrate were representative of experimental values because of a lack of data. However, because the recycle system was a conservative system (i.e., constituents were not lost by washout) it was likely that the actual mass of solid substrate removed in the simulation was a good approximation of the amount removed experimentally during the acid phase if the amount of volatile acids produced in the simulation closely approximated the amount of volatile acids produced experimentally. In other words, if all hydrolysis products produced from solid substrate were converted to volatile acids, and the simulation of volatile acid concentration closely approximated the experimental volatile acids by methanogenesis to complicate the mass balance), then it can be deduced that the amount of solid substrate removed by hydrolysis/solubilization in the simulation was representative of experimental removal of solid substrate. The terms "mass of volatile acid" and "volatile acid concentration" can be used interchangeably only when flows are accurately known so that the conversion from concentration to mass is accurate.

The acidogen concentration profile for the recycle simulation was virtually identical to the single-pass simulation (Figure 14) as was the hydrolysis product concentration profile (Figure 15). This similarity indicated that the initial concentration of hydrolysis products was determined by the peak in acidogen concentration and that initial removal of hydrolysis products was largely due to acidogenesis rather than to washout in the single-pass simulation.

Differences between the recycle and single-pass simulations were largely manifest through methanogen concentration, volatile acid concentration, and methane production profiles. Methanogen concentration for the recycle simulation (Figure 16) initiated at level of 10 mg/L on Day 200, increased to a peak value of 670 mg/L on Day 366, then decreased thereafter. This was in sharp contrast to the single-pass simulation which never attained a methanogen concentration higher than the initial value. These differences can be attributed to the fact that most of the volatile acids in the single-pass simulation were removed by washout and were thus unavailable for

FIGURE 13

SOLID SUBSTRATE CONCENTRATION IN LANDFILL MOISTURE, RECYCLE SIMULATION

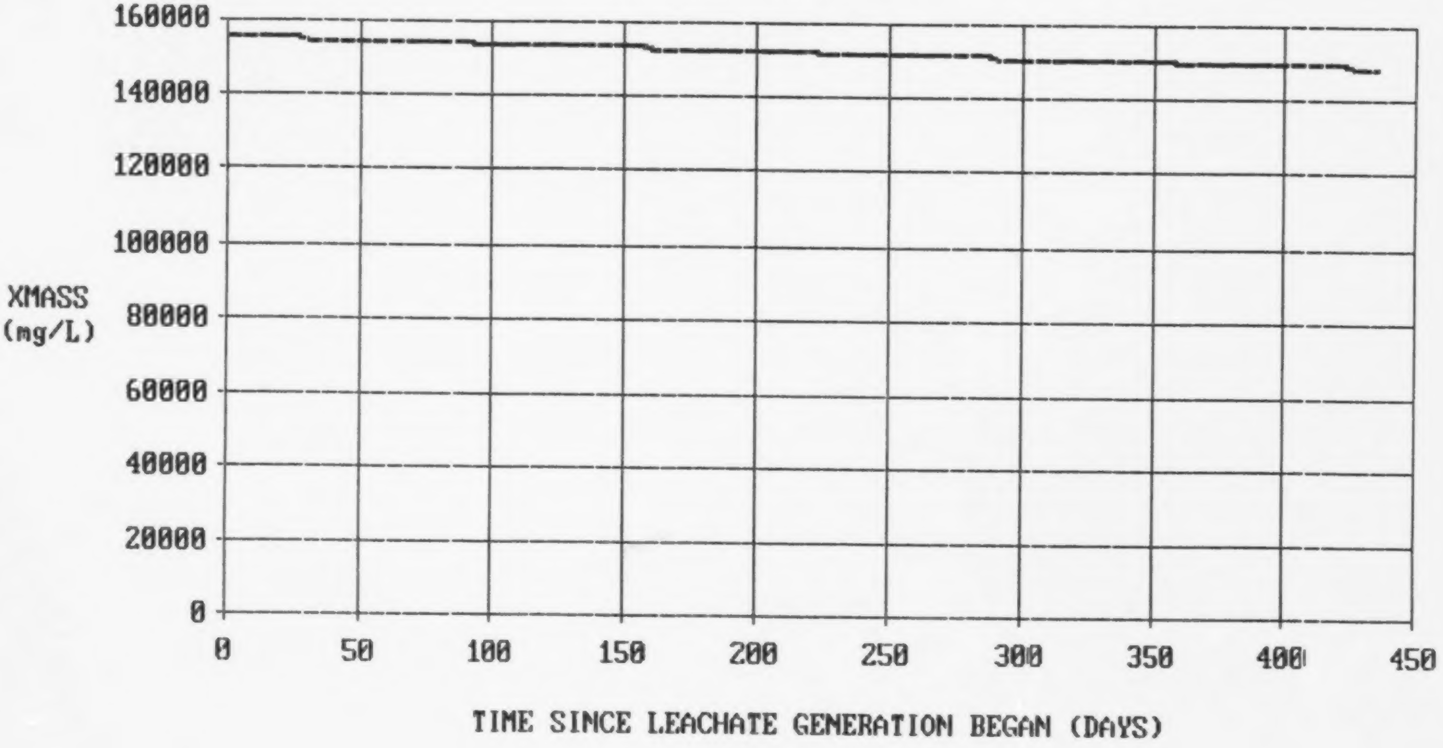


FIGURE 14

ACIDOGEN CONCENTRATION IN LANDFILL MOISTURE,
RECYCLE SIMULATION

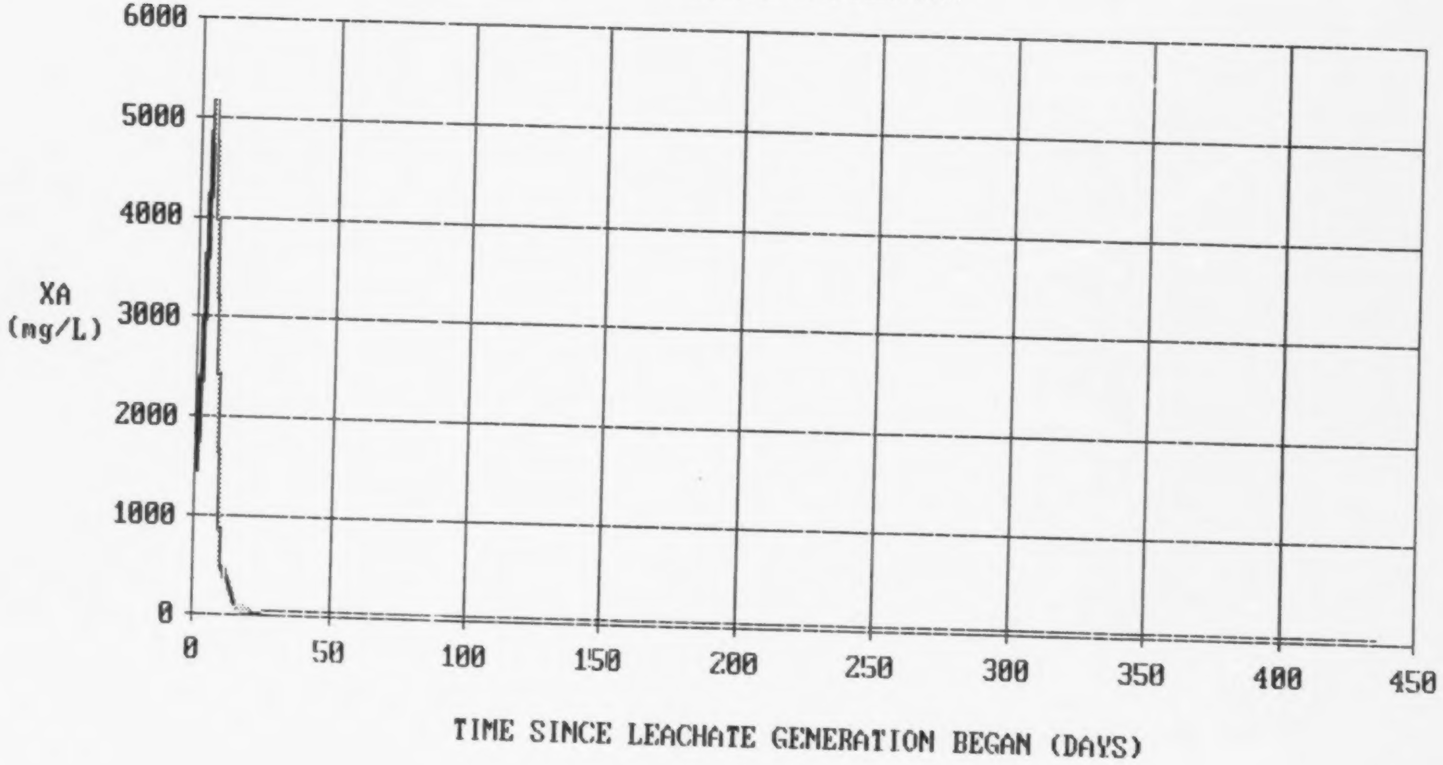


FIGURE 15

HYDROLYSIS PRODUCT CONCENTRATION IN LANDFILL LEACHATE,
RECYCLE SIMULATION

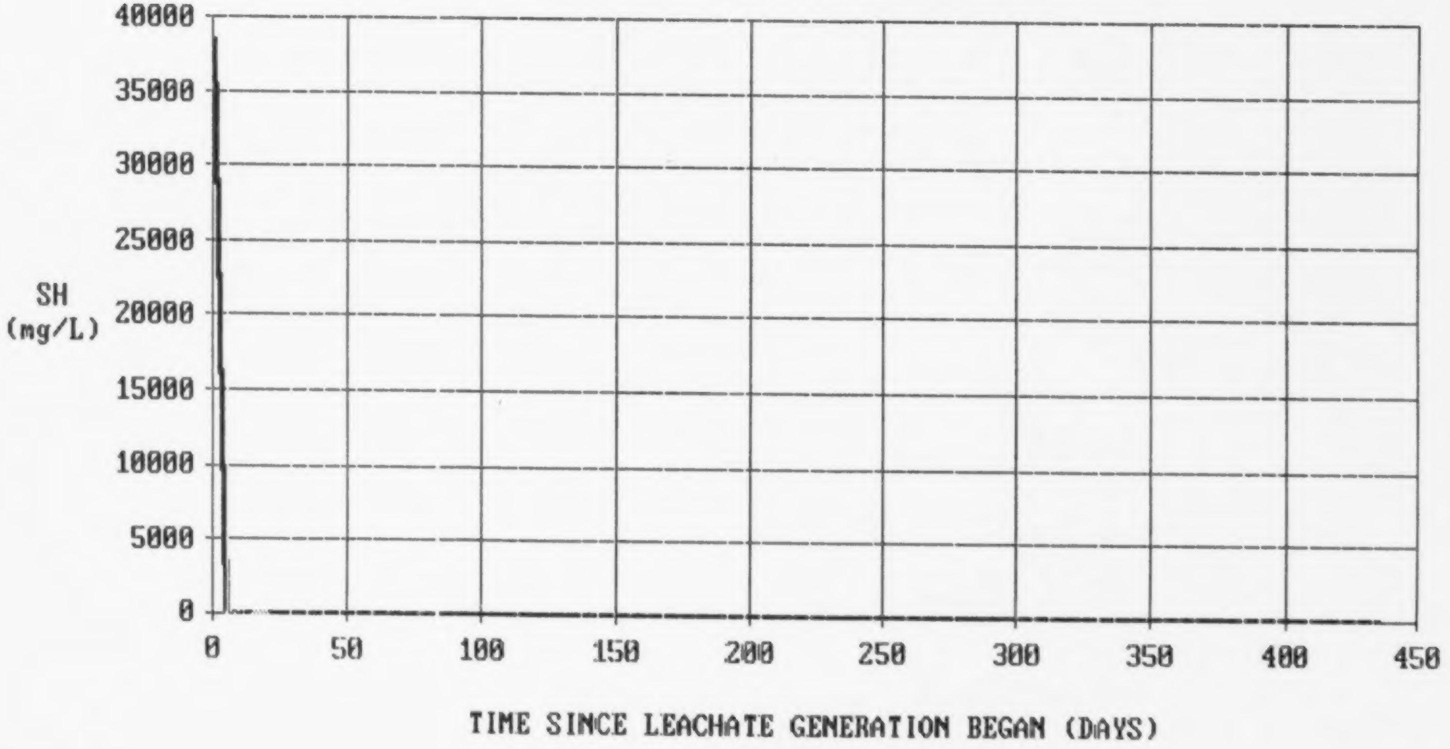


FIGURE 16

METHANOGEN CONCENTRATION IN LANDFILL MOISTURE, RECYCLE SIMULATION

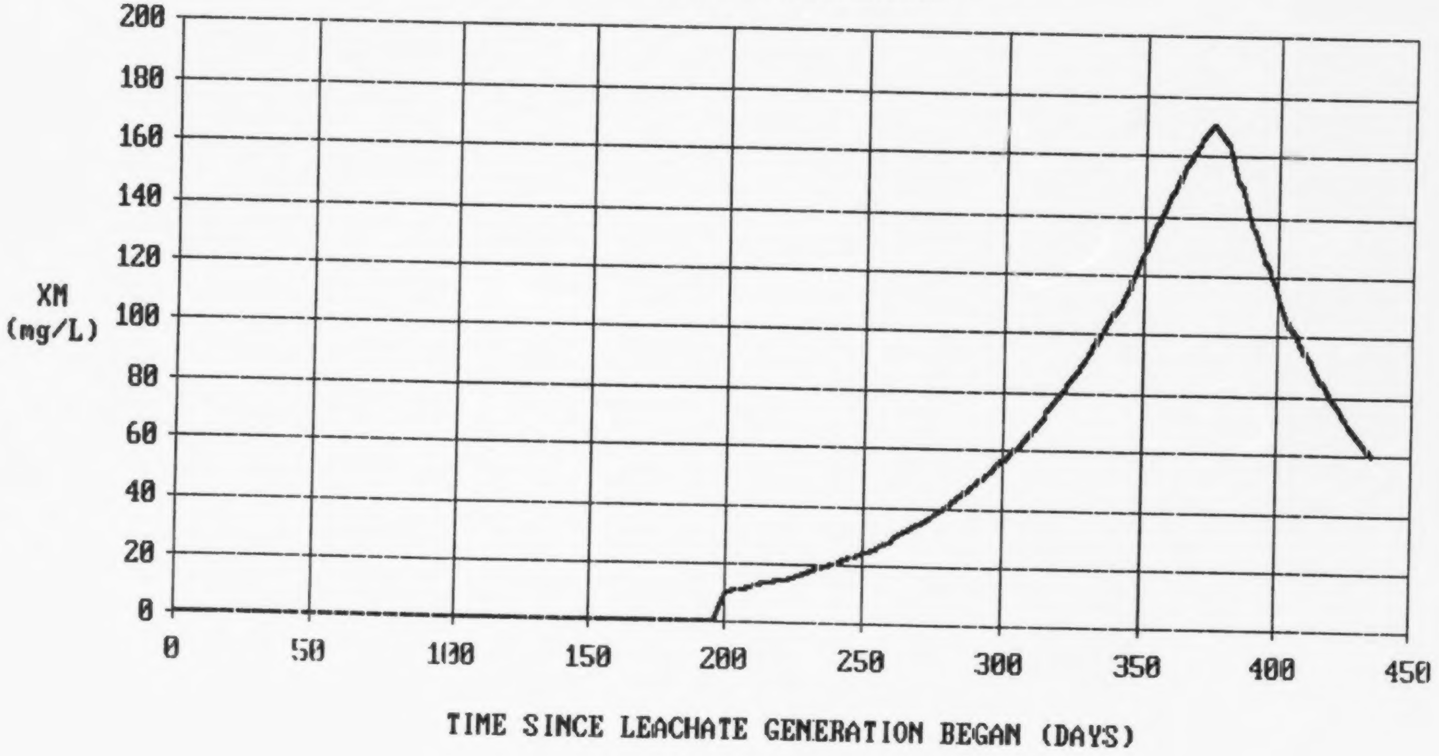


FIGURE 17

VOLATILE ACID CONCENTRATION IN LANDFILL LEACHATE, RECYCLE SIMULATION

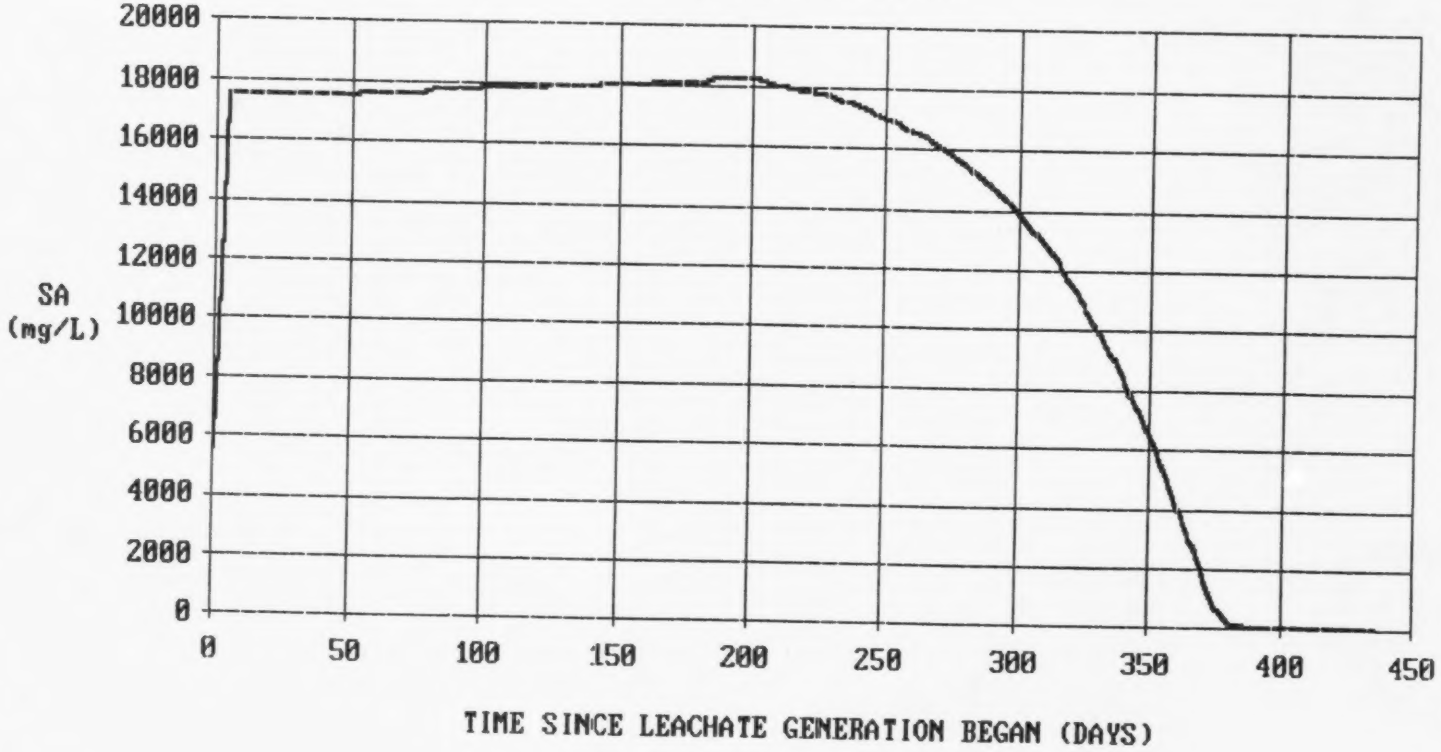


FIGURE 18

COD CONCENTRATION IN LANDFILL LEACHATE,
RECYCLE SIMULATION

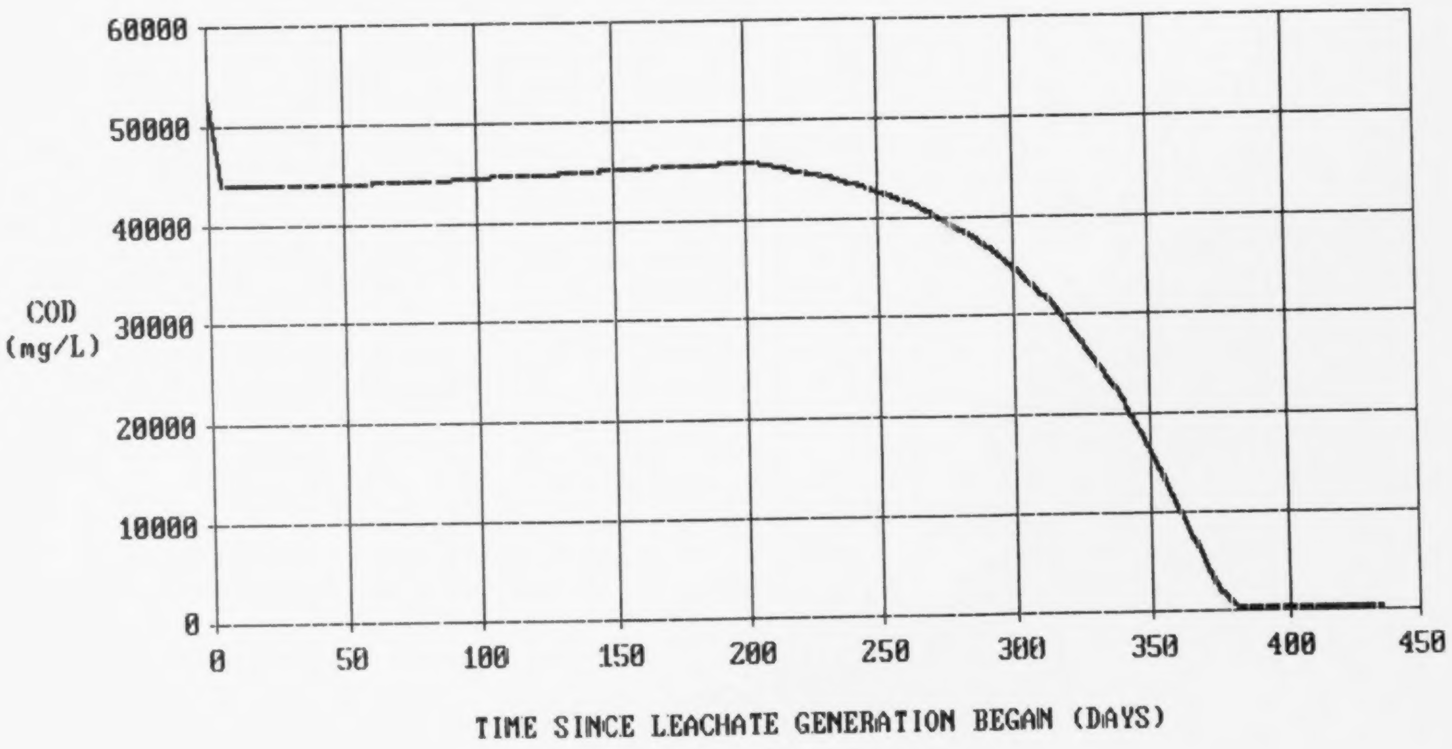
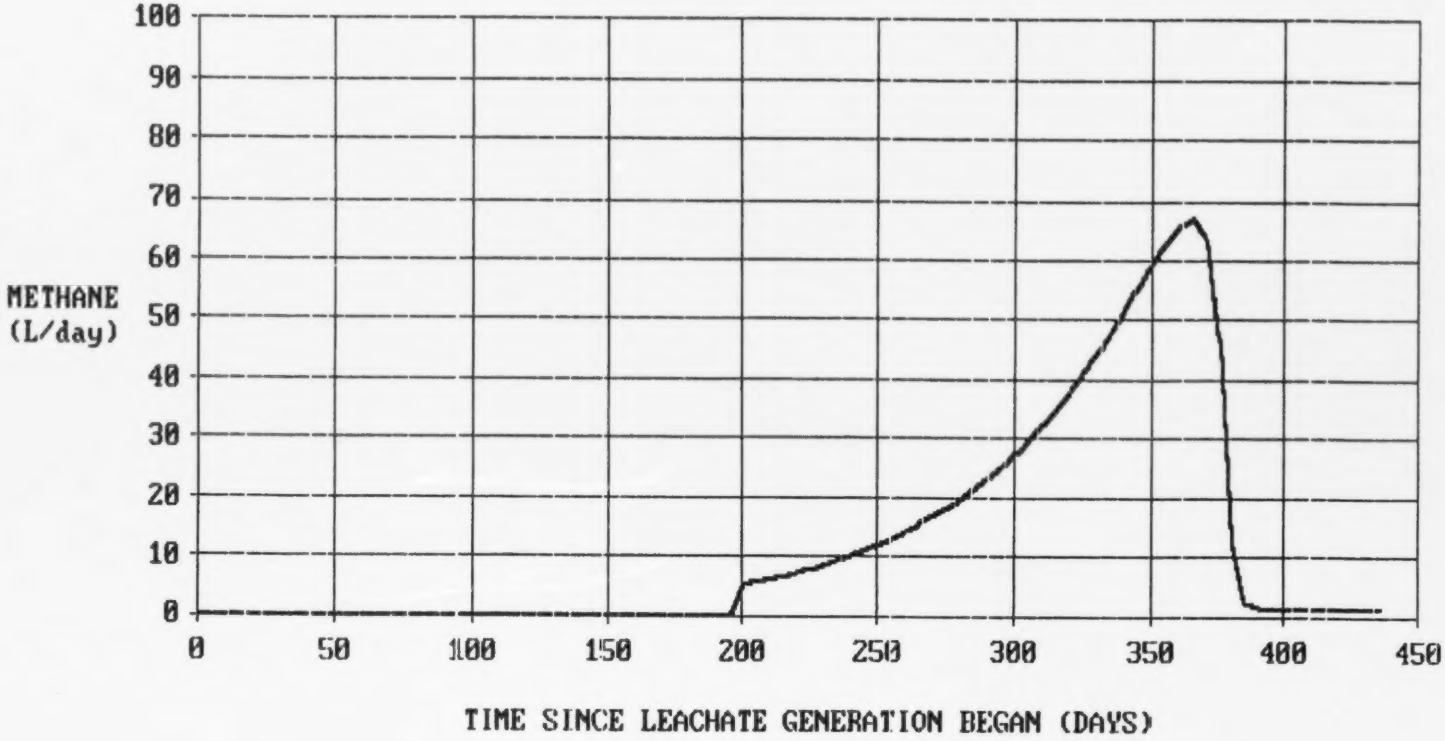


FIGURE 19

METHANE PRODUCTION,
RECYCLE SIMULATION



utilization by methanogens, whereas all volatile acids removed in the recycle system were removed by methanogenesis.

The volatile acid concentration profile for the recycle simulation (Figure 17) was also markedly different from the single-pass simulation (Figure 10). The volatile acid concentration increased from the initial value of 5,000 mg/L to 17,5000 mg/L within the first week and continued to increase gradually until Day 200. On Day 200, corresponding with initiation of methanogenesis, the volatile acid concentration began to decrease rapidly, falling below 10 mg/L by Day 390. Volatile acid concentration then gradually trended upward for the remainder of the recycle simulation. The COD concentration profile (Figure 18) was again very similar in shape to the volatile acid concentration profile with the exception of the initial peak.

Methane production in the recycle simulation (Figure 19) followed the trend of methanogen growth. Methane production increased rapidly, peaking at 670 L/day on Day 366, then decreased more rapidly than the decrease in methanogen concentration, falling to a value of 2 L/day by Day 368 as the concentration of volatile acids was depleted.

Figures 6 through 19 have been presented as a means to convey the mechanics of GTLEACH-I. However, most of the parameters output by GTLEACH-I are seldom measured in landfill experiments. For this reason fitting of the model to experimental data must be limited to those parameters which are measured. Of those parameters output by GTLEACH-I, available experimental data are typically limited to COD, volatile acid concentration and methane production.

Although COD is the most frequently measured leachate parameter in landfill experiments, it can be the most unreliable of the three due to losses of volatile acids by volatilization during the COD digestion procedure. Not only is COD likely to be inaccurate in terms of other commonly available leachate characteristics measured during landfill experiments, it can also be misleading in its interpretation since COD analysis measures not only COD contributed by hydrolysis products and volatile acids, but also other constituents of leachate such as extracellular enzymes, microbial material and refractory components of the solid waste. Thus it was difficult to compare experimentally measured COD with the sum of COD contributions of simulated volatile acids and hydrolysis products.

Volatile acid concentration in leachate can be accurately measured by gas chromatographic techniques. Since volatile acids are produced in landfill leachate almost solely as a result of microbial activity, volatile acid concentration could be directly related to the processes simulated by GTLEACH-I. In fact, measurement of volatile acid concentration in leachate allows monitoring of the net effects of three of the processes occurring in the landfill: production of volatile acids by acidogens, removal of volatile acids by methanogens, and removal of volatile acids by washout. Thus volatile acid concentration was a key parameter in fitting GTLEACH-I to experimental data.

As long as accurate gas volume measurements are made and there are no gas leaks in landfill experiments, the methane fraction of landfill gases can be accurately measured by gas partitioning and converted to volume. Measurements of methane production from landfills allows monitoring of the activity of methanogens.

The greatest difficulty in verifying the three-step microbial model is the frequent lack of any direct experimental measurement of hydrolysis product concentration. Without direct measurement, accurate modeling of hydrolysis product concentration can only be inferred from correlation of simulated and experimental volatile acid concentration and methane production.

SECTION 7

FITTING GTLEACH-I TO EXPERIMENTAL DATA

The GTLEACH-I model has been applied to the simulation of a landfill experiment begun by Schaffer (1986) and completed by Yari (1986). In the experiment, two pilot-scale (208-L) landfill cells were filled with shredded municipal solid waste (MSW). One of the cells was operated in a leachate recycle mode while the other was operated in single-pass mode.

The cells were each packed with 82 kg of shredded MSW having a moisture content of 33%. The MSW absorbed an additional 44 liters of applied water before attaining field capacity. Because the waste was shredded, actual determination of the food fraction was not possible, so it was assumed that the refuse contained 20% food waste or degradable waste on a mass weight basis (Tchobanoglous et al., 1977). Thus the GTLEACH-I input parameters XMASSO, the total mass of degradable solid substrate, and TVOL, the field capacity of the cell, were set at 11 kg and 71 liters, respectively.

The rate of moisture application for the single-pass cell was 6 L/wk once field capacity was attained (equivalent to 127 cm/yr of rainfall with 100% infiltration). The average daily flow, F, was then 0.857 liters. For the recycle cell, approximately 25 L/wk of leachate were recycled through the cell, giving an average daily flow of 3.6 liters. These high rates of moisture application were designed to provide accelerated stabilization of the municipal solid waste, but also to delay the onset of the methane fermentation phase intentionally imposed on both the recycle and single-pass cells. These delays in natural methanogenesis activity prompted eventual seeding of the cells with digested sludge and development of methanogenesis activity after 200 days of leachate production. Therefore, a time lag of 200 days was selected before initiation of the methanogen population in this simulation.

SIMULTANEOUS FITTING

Fitting of the GTLEACH-I to the Schaffer (1986) and Yari (1986) data was initiated by first selecting values for the kinetic constants. Values of kinetic growth constants for acidogens and methanogenic bacteria compiled by Henze and Harremoes (1983) are included in Tables 1 and 2. The tables present values for maximum specific growth rates, u_{max}, rather than maximum substrate utilization rates, Q. Maximum substrate utilization rates can be obtained by dividing maximum growth rate by a corresponding yield coefficient. Median values of kinetic growth constants shown in the tables were selected as initial values from which to begin the fitting procedures.

Few literature values are available for hydrolysis rate constants. Eastman and Ferguson (1981) presented a value of 0.3/day at 35°C for a first-order hydrolysis rate constant measured in a separate acid-producing reactor fed with raw domestic primary sludge. The hydrolysis rate constant measured in Eastman and Ferguson's (1981) experiment with primary sludge would not necessarily be expected to apply to the hydrolysis of shredded municipal solid waste. Large particles with low surface-to-volume ratios such as shredded MSW would be hydrolyzed more slowly than smaller particles such as those resulting in primary sludge fed to completely mixed digestors. Starches, proteins and cellulose are degraded at different rates. In addition, the overall first-order hydrolysis rate constant represents the sum of individual processes occurring in the landfill system and would reflect the relative composition of the substrate. Therefore, the initial value selected for K_H was based on an order-of-magnitude approximation of the Eastman and Ferguson (1981) value. Parameters selected for initial simulation were:

Flow, F = 6 L/wk = 0.86 L/day
Moisture content, TVOL = 71 L,
Number of CSTRs in series, NCSTR = 3
Initial mass of solid substrate, XMASSO = 11 kg,
Initial concentration of hydrolysis products, SHO = 40,000 mg/L,
Initial concentration of volatile acids, SAO = 5,000 mg/L,
Initial concentration of acidogens, XAO = 100 mg/L, and
Initial concentration of methanogens, XMO = 10 mg/L.

Kinetic parameters for the various processes were:

Hydrolysis	Acidogenesis	Methanogenesis
K_H = 0.1/day	Q_A = 3.5/day	Q_M = 1.5/day
	K_A = 200 mg/L	K_M = 500 mg/L
	K_{DA} = 0.5/day	K_{DM} = 0.02/day
	Y_A = 0.5	Y_M = 0.02
		Methanogen lag = 200 days

A preliminary simulation was conducted using these values. Simulated results of volatile acid concentration as a function of time were compared with experimental results. The parameters were then adjusted to fit the experimental results of volatile acid concentrations with time in subsequent simulations.

Several objectives were applied as a first attempt at fitting the parameters. An effort was made during parameter adjustment to fit both recycle and single-pass volatile acid data simultaneously with more emphasis placed on the recycle cell as it provided a clearer indication of the impact of microbial processes without washout effects. Although microbial populations in the two types of reactors were not expected to have exactly the same growth rate parameters, it was decided that an initial simultaneous fitting would provide a set of parameters from which to explore the differences in kinetic constants imposed by the two leachate management options. A second initial objective was to avoid underestimating the peak volatile acid concentration observed in the data, as this represented the most aggressive leachate produced with the potentially greatest environmental impact. Gross fitting of the model was accomplished by adjusting the maximum rate constants; K_H, Q_A, and Q_M, for these parameters were expected to show the largest deviation from literature values. Fine-tuning was then conducted with small adjustments of the half-velocity constants, decay rates and yield coefficients. In accordance with the above objectives, the following group of kinetic parameters were finally selected as producing the best fit for both recycle and single-pass data simultaneously:

Hydrolysis	Acidogenesis	Methanogenesis
K_H = 0.0001/day	Q_A = 3.2/day	Q_M = 1.9/day
	K_A = 200 mg/L	K_M = 500 mg/L
	K_{DA} = 0.5/day	K_{DM} = 0.02/day
	Y_A = 1.0	Y_M = 0.02
		Methanogen lay = 200 days

The only major deviation in the fitted parameters from literature values was the hydrolysis rate constant which was three orders of magnitude lower than the literature value reported by Eastman and Ferguson (1981). That the simulated value of hydrolysis rate constant differed from the Eastman and Ferguson (1981) value was not surprising considering the previous discussion of factors influencing hydrolysis rate constants. However, the degree of deviation from the Eastman and Ferguson (1981) value is cause for question and is examined subsequently.

The fitting value of maximum substrate utilization rate for acidogens of 3.2/day was at the low end of the range provided in Table 1. The fitted half-velocity constant for acidogens was consistent with experimental values reported in Table 1 as was the decay rate for acidogens. The value of the yield coefficient of 1.0 for acidogens was approximately twice the typical values reported in Table 1.

The maximum substrate utilization rate for methanogens of 1.9/day was somewhat higher than the rates measured in acetate enrichment experiments reported by Ghosh and Klass (1978) shown in Table 2. For many of the experiments reported in Table 2, however, the maximum substrate utilization rate for methanogens could not be determined due to insufficient information (either no maximum specific growth rate or no yield coefficient). The half-velocity constant for methanogens resulting from fitting of GTLEACH-I was well within the range of values reported in Table 2 as were the decay rate and yield coefficient.

TABLE 1

GROWTH CONSTANTS OF ACID PRODUCING ANAEROBIC BACTERIA

Maximum specific rate	**Yield coefficient**	**Half velocity constant**	**Substrate removal rate**	**Decay rate**	**Temperature**	**Culture/ substrate**	**Literature**
μ_{max}	Y_A	K_A	r_x	K_{DA}			
d^{-1}	kg VSS/ kg COD	mg COD/L m^3	kg/COD/ (kg VSS˙d)	d^{-1}	°C		
30	0.17	23		6.1	37	Mixed anaerobid (dextrose)	Ghosh and Pohland (1974)
	0.34			0.43	35	Acid producing sludge	Eastman and Ferguson (1981)
		250-3200	0.88			Mixed anaerobic (model)	Lindgren (1982)
		150	1			Mixed anaerobic	Mueller and Mancini (1975)
		2000	2.2				
	0.12			0.08		Mixed anaerobic	Young and McCarty (1967)
3.8	0.40	37000		0.79	37	Acid producing sludge (sludge feed	Ghosh et al. (1975)
>1.33	0.54			0.87	38	Mixed	Andrews and Pearson (1965)
	0.41			0.10			Speece and McCarty (1964)
	0.26					Theoretical (glucose)	Sykes (1975)
	0.15	192	7.5		35	Theoretical/ Model (glucose)	McCarty (1971)
2.7					33	Mixed (design)	Gujer and Zehnder (1982)
7.2	0.14	370			35	Glucose enrichm.	Ghosh and Klass (1978)
3.8	0.28	18300			35	Sewage sludge	Ghosh and Klass (1978)
1.7	0.13	30900			35	Cellulose enrichm	Ghosh and Klass (1978)

TABLE 2

GROWTH CONSTANTS OF METHANE PRODUCING ANAEROBIC BACTERIA

Maximum specific rate	Yield coefficient	Half velocity constant	Substrate (HAc or HAc Equiv.) removal rate	Decay rate	Temperature	Culture/ remark	Literature
μ_{max}	Y_M	K_M	r_x	K_{DM}			
d^{-1}	kg VSS/ kg COD	mg HAc/L	kg/HAc/ (kg VSS·d)	d^{-1}	°C		
		2.0	4			Mixed anaerobic	Mueller and Mancini (1975)
0.08-0.09	0.02		2.4-4.8		35	Acetate enrichment	van den Berg (1977)
	0.02		2.4-4.8		30	"	" " " "
	0.05		2.4		20	"	" " " "
	0.05			0.04		Mixed anaerobic	Young and McCarty (1976)
0.34	0.04	160.	8.2	0.02	35	Acetate enrichment	Lawrence and McCarty (1969)
		19			33	Anaerobic sludge	Kaspar (1977)
	0.045-0.055	200	4.5-7.5	0.01-0.02	30	Model	Hartmann (1981)
0.26		10			30	Acetate enrichment	Cappenberg (1975)
3.4		600			37	Anaerobic sludge	Ghosh and Pohland (1974)
		8 80				Mixed anaerobic/ model	Lindgren (1982)
	0.04	160		0.02	35	(Model)	Mosey (1983)
		950	0.4		25	(Fixed bed/model)	Switzenbaum (1982)
		1000	0.8		35	(Exp.bed/model)	" "
			10-12		35	Acetate enrichment	Speece et al. (1982)
	0.03	60	7.5		35	(Theoretical/ model)	McCarty (1971)

TABLE 2 (continued)

GROWTH CONSTANTS OF METHANE PRODUCING ANAEROBIC BACTERIA

Maximum specific rate	Yield coeffi-cient	Half velocity constant	Substrate (HAc or HAc Equiv.) removal rate	Decay rate	Tempe-rature	Culture/ remarks	Literature
μ_{max}	Y_M	K_M	r_x	K_{DM}			
d^{-1}	kg VSS/ kg COD	mg HAc/L m	kg/HAc/ (kg VSS·d)	d^{-1}	°C		
	(0.039)					(Theoretical)	Bauchop and Elsden (1960)
0.4	0.06	2				(Model/Mixed culture	Andrews and Graef (1971)
	0.04- 0.05	154	8.1	0.01- 0.04	35		Reference Unavailable
		333	4.8		30		" "
		869	4.7		25		" "
0.24	0.04	70		0.02		(Model)	Reference Unavailable
	0.07- 0.09		(0.6- 0.9)		30	Mixed (soured sugar-beet waste)	Lettinga et al. (1980a)
	0.03- 0.04		(0.7- 0.9)		30		" " "
1.4		300			50	Pure culture	Zinder and Mah (1979)
0.5-0.7	0.03- 0.04	300			36	Pure culture	Smith and Mah (1978)
			2.1-2.2		30	Acetate enrichm.	Pol et al. (1982)
			3.7		38	Acetate enrichm.	" " " "
>1.33	0.14			0.02	38	Mixed	Andrews and Pearson (1965)
0.3		200			33	Mixed (design)	Guger and Zehnder (1982)
0.49	0.28	3900			35	Acetate enrichm.	Ghosh and Klass (1978)

A graphical display of the comparison between the experimental total acids data and the simulated results are shown in Figures 20 and 21. The corresponding results for COD are also shown in Figures 22 and 23 and for methane production in Figures 24 and 25. It should be pointed out that this simulation was the same as that presented previously. As a first trial, the shapes and magnitudes of the simulated curves showed reasonable correspondence with the data. There were, however, several important deviations from the experimental data.

For this single-pass simulation, the entire simulated profile of volatile acid concentration was shifted approximately 20 days earlier than the experimental data. Furthermore, the simulated volatile acid curve underestimated the concentration at which the experimental data leveled out. Both these deviations were manifest to a lesser degree in the COD simulation. Examination of methane production indicated that the initial concentration of methanogens (10 mg/L) was too high and needed to be lowered to allow for an increase in methane production to the peak value (Figure 9).

For the recycle simulation, the criteria which required that peak volatile acid concentrations not be underestimated resulted in poorer fitting of the volatile acid data between Days 20 and 250 than might otherwise have been obtained (also shown in the COD curve). The methane production simulation for the recycle mode showed good correspondence with the data, although the initial value of methanogen concentration was too high in this case as was evidenced by the abrupt initiation of methane production.

Before further discussion of reasons for the preceding deviations from experimental data and exploration of remedies which could be made, a sensitivity analysis of the volatile acid profiles performed on the fitted parameters was considered instructive.

SENSITIVITY ANALYSIS

By selection of a range of values for the various model parameters, the impact of each value on model output was examined. As demonstrated in Figures 26 and 27, varying the number of CSTR's in series from 1 to 5 had only a small effect on the single-pass simulation and virtually none on the recycle simulation. The effect of increasing the number of CSTR's was to shift the washout curve of the volatile acids from one resembling a tracer study for a completely mixed reactor to one resembling a plug flow reactor. Since there was no washout in the recycle system, the sole removal mechanism for volatile acids was by microbial degradation which was not significantly affected by the mixing regime.

The results shown in Figures 28 and 29 demonstrate the effects of adjustments to the value of the first-order rate constant for hydrolysis/solubilization, K_H. Increasing the value of K_H from 0.00005/day to 0.0005/day shifted both the recycle and single-pass simulated volatile acid curves upward, with a greater change seen in the recycle simulation. The effect served to slightly improve the fit on the single-pass data, but produced a negative effect on the fit of the recycle data. This increase in

FIGURE 20

COMPARISON OF SIMULATED VOLATILE ACID CONCENTRATION WITH EXPERIMENTAL DATA (SCHAFFER 1986, YARI 1986), SINGLE-PASS

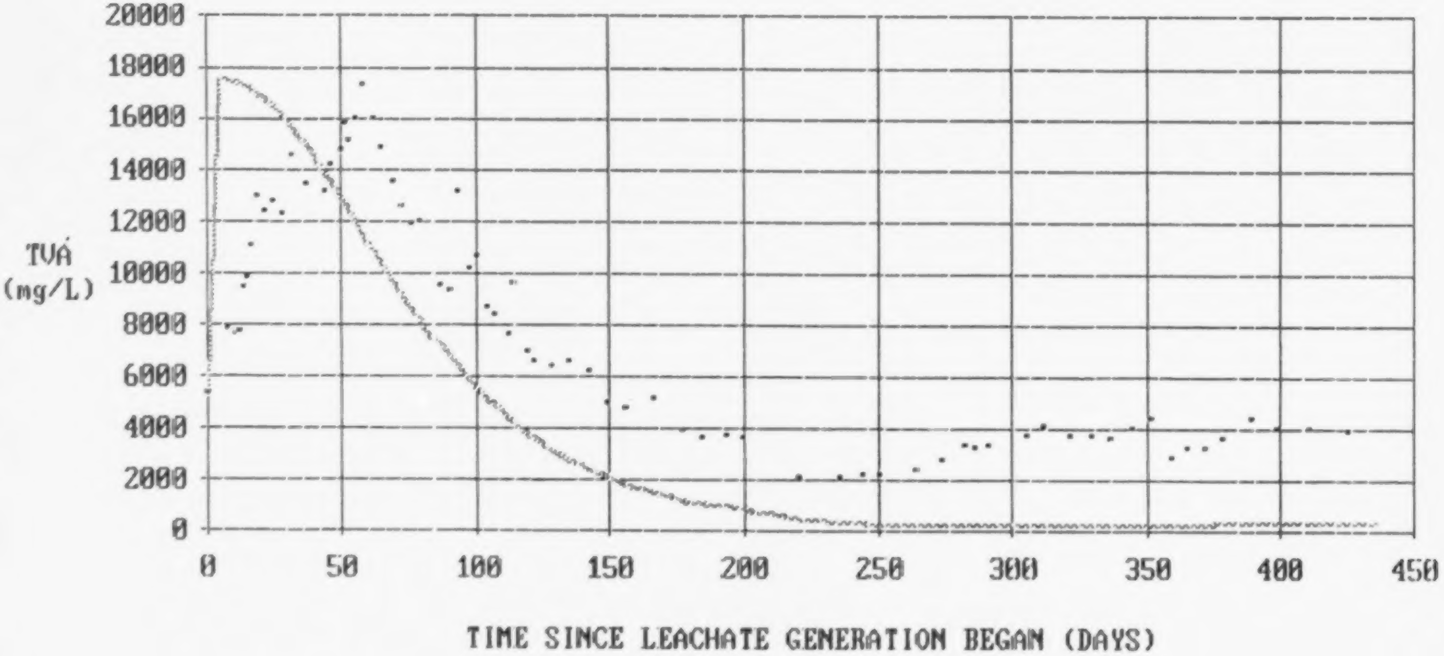


FIGURE 21

COMPARISON OF SIMULATED VOLATILE ACID CONCENTRATION WITH EXPERIMENTAL DATA (SCHAFFER 1986, YARI 1986), RECYCLE

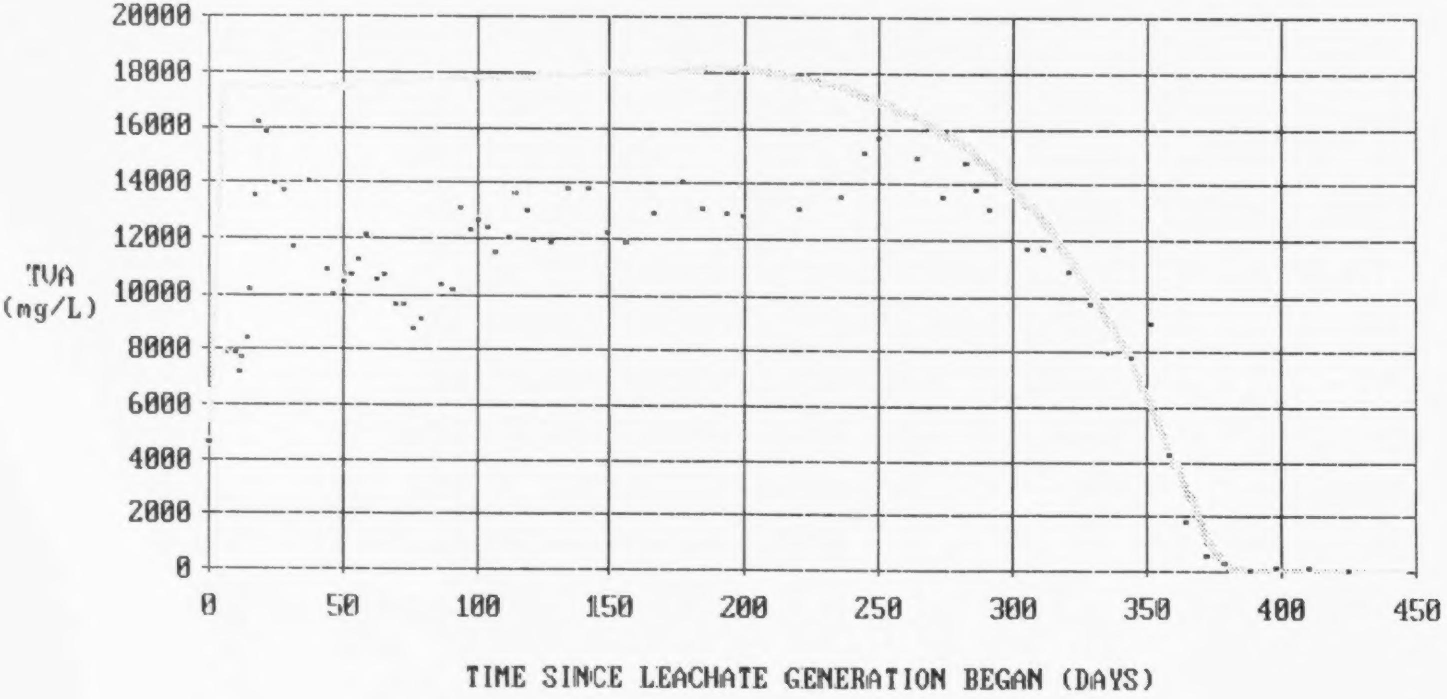


FIGURE 22

COMPARISON OF SIMULATED COD WITH EXPERIMENTAL DATA
(SCHAFFER 1986, YARI 1986), SINGLE-PASS

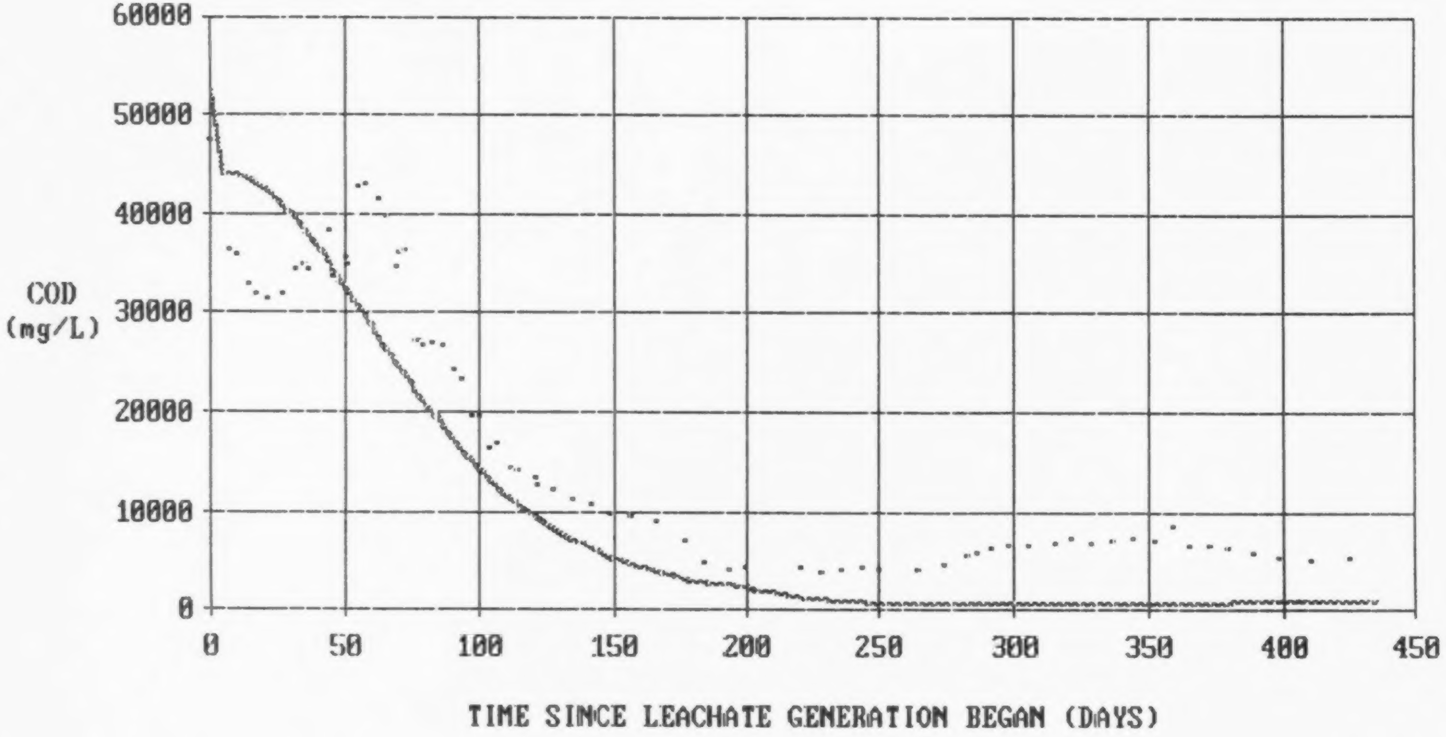


FIGURE 23

COMPARISON OF SIMULATED COD WITH EXPERIMENTAL DATA
(SCHAFFER 1986, YARI 1986), RECYCLE

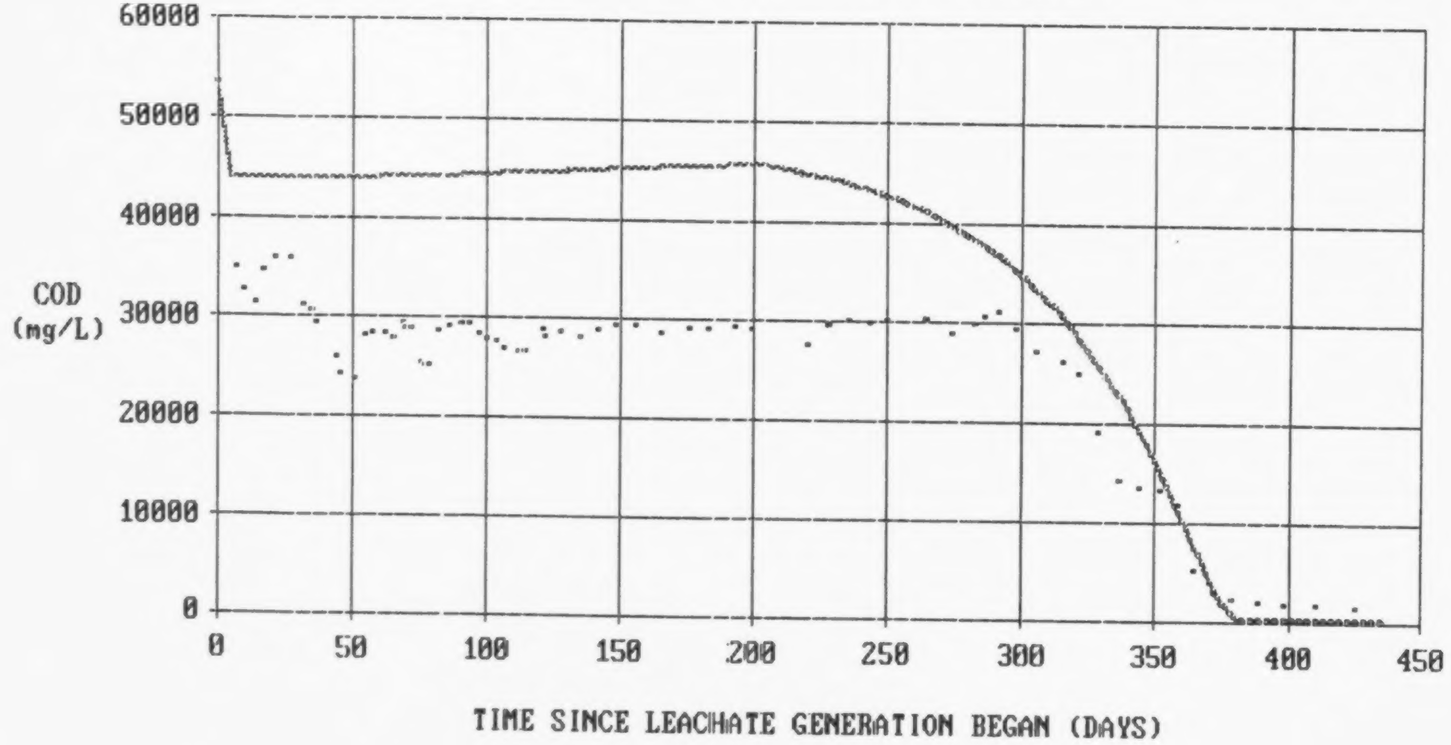


FIGURE 24

COMPARISON OF SIMULATED METHANE PRODUCTION WITH EXPERIMENTAL DATA
(SCHAFFER 1986, YARI 1986), SINGLE-PASS

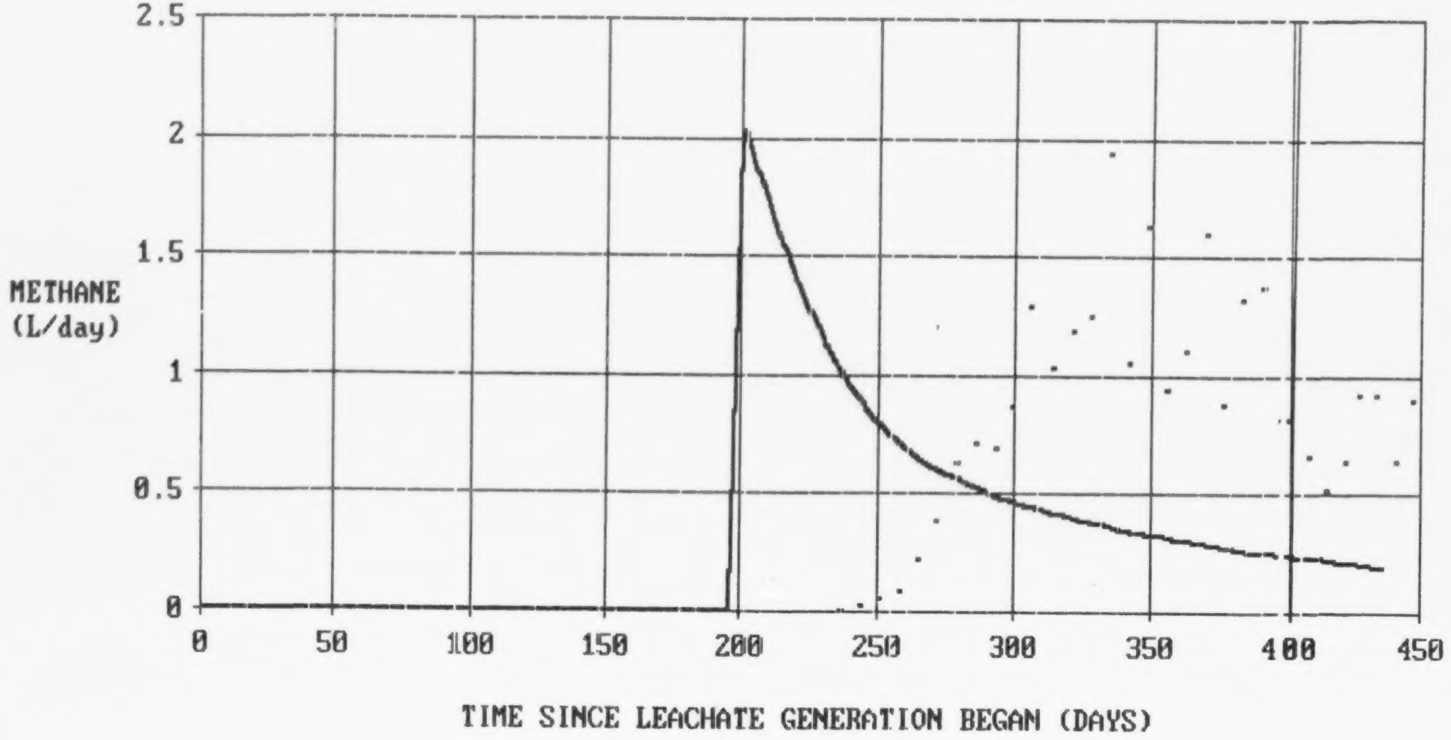


FIGURE 25

COMPARISON OF SIMULATED METHANE PRODUCTION WITH EXPERIMENTAL DATA
(SCHAFFER 1986, YARI 1986), RECYCLE

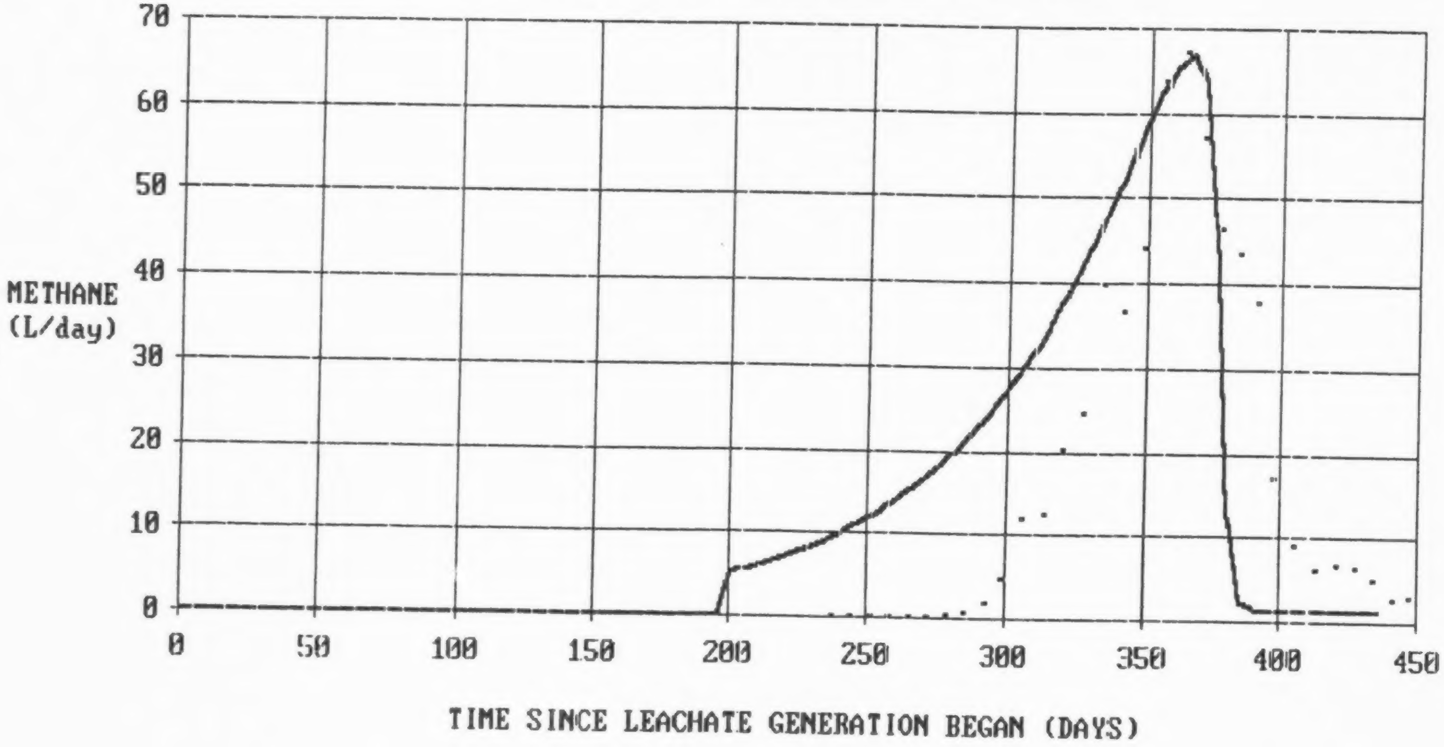


FIGURE 26

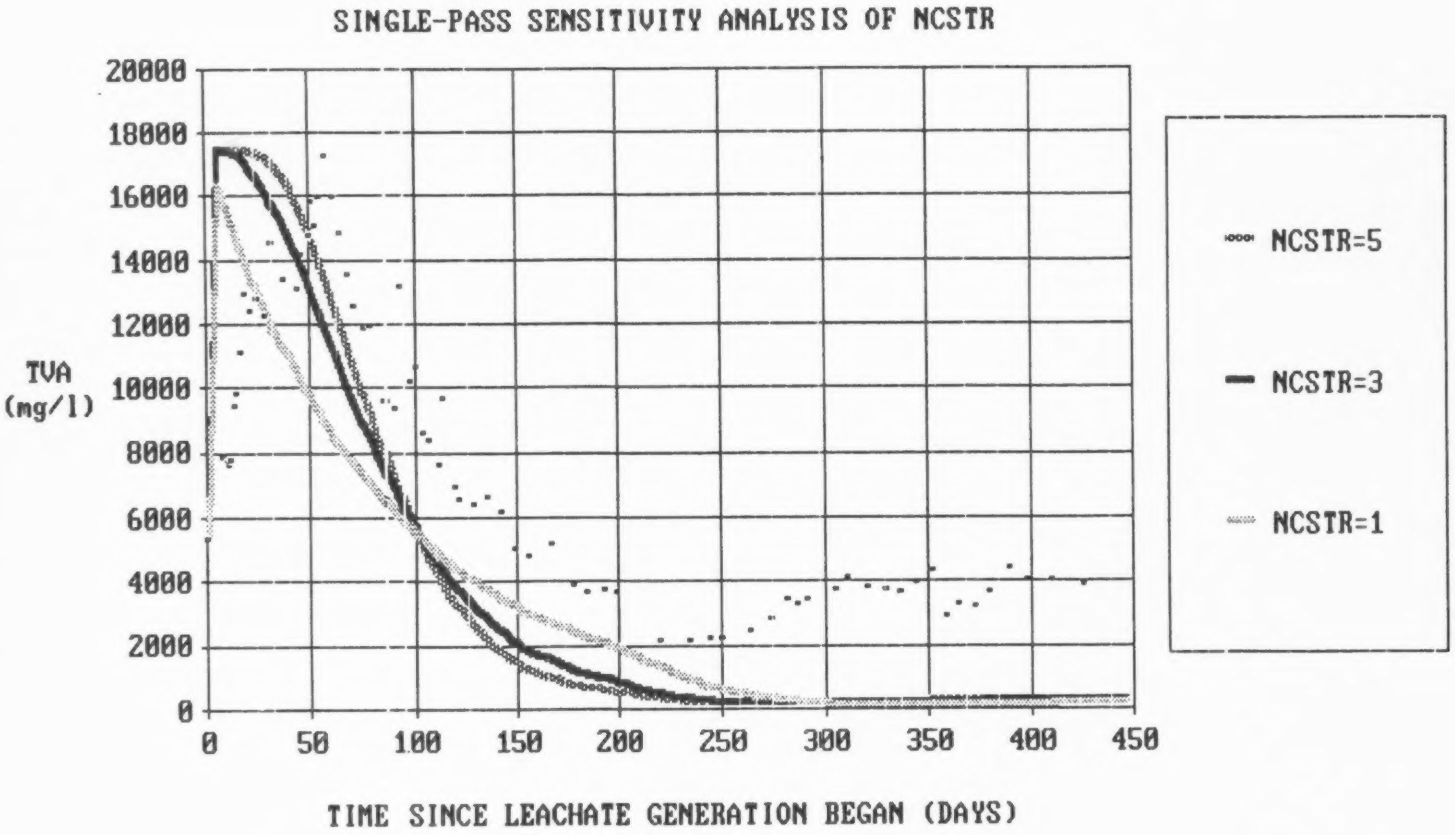

SINGLE-PASS SENSITIVITY ANALYSIS OF NCSTR

FIGURE 27

RECYCLE SENSITIVITY ANALYSIS OF NCSTR

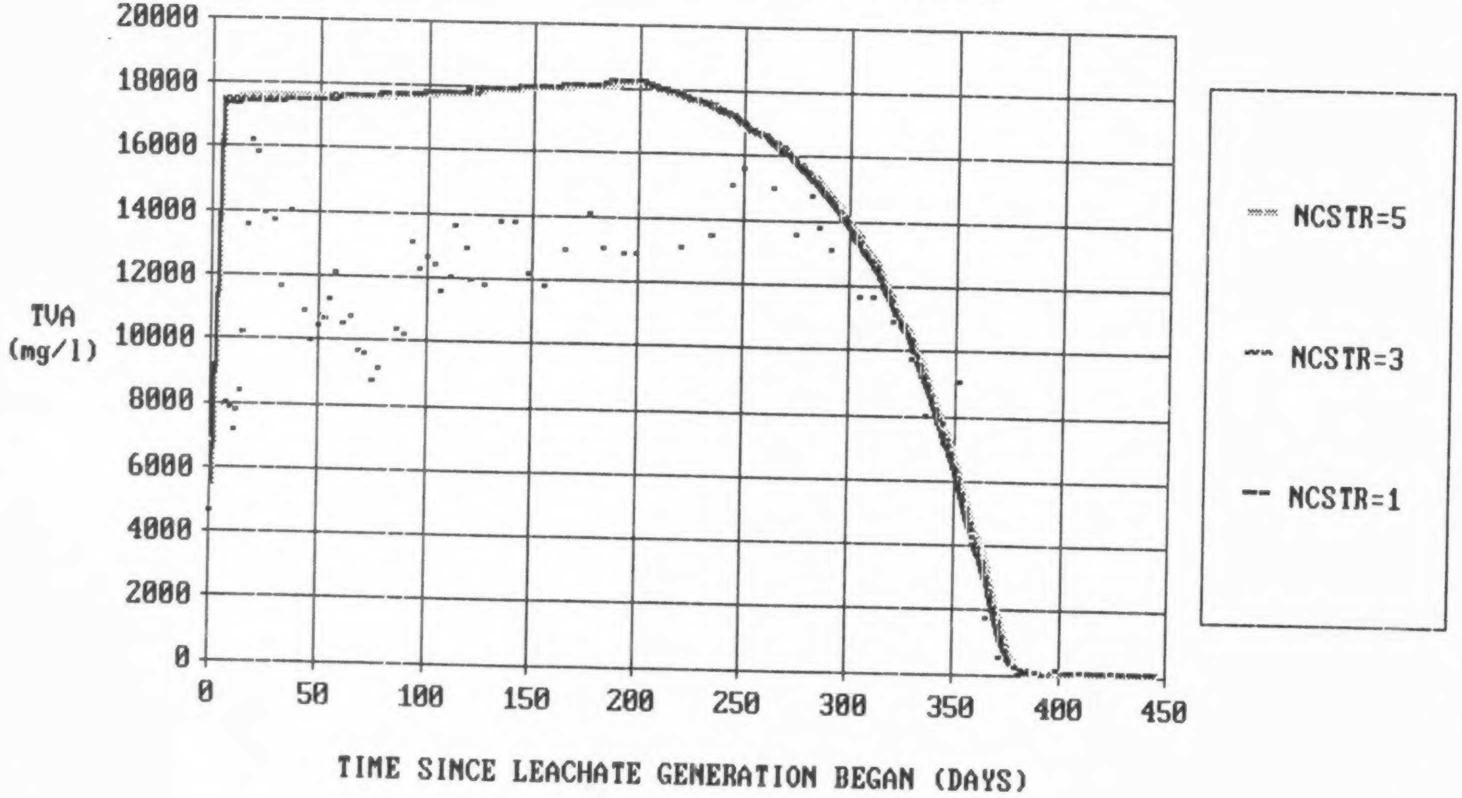


FIGURE 28

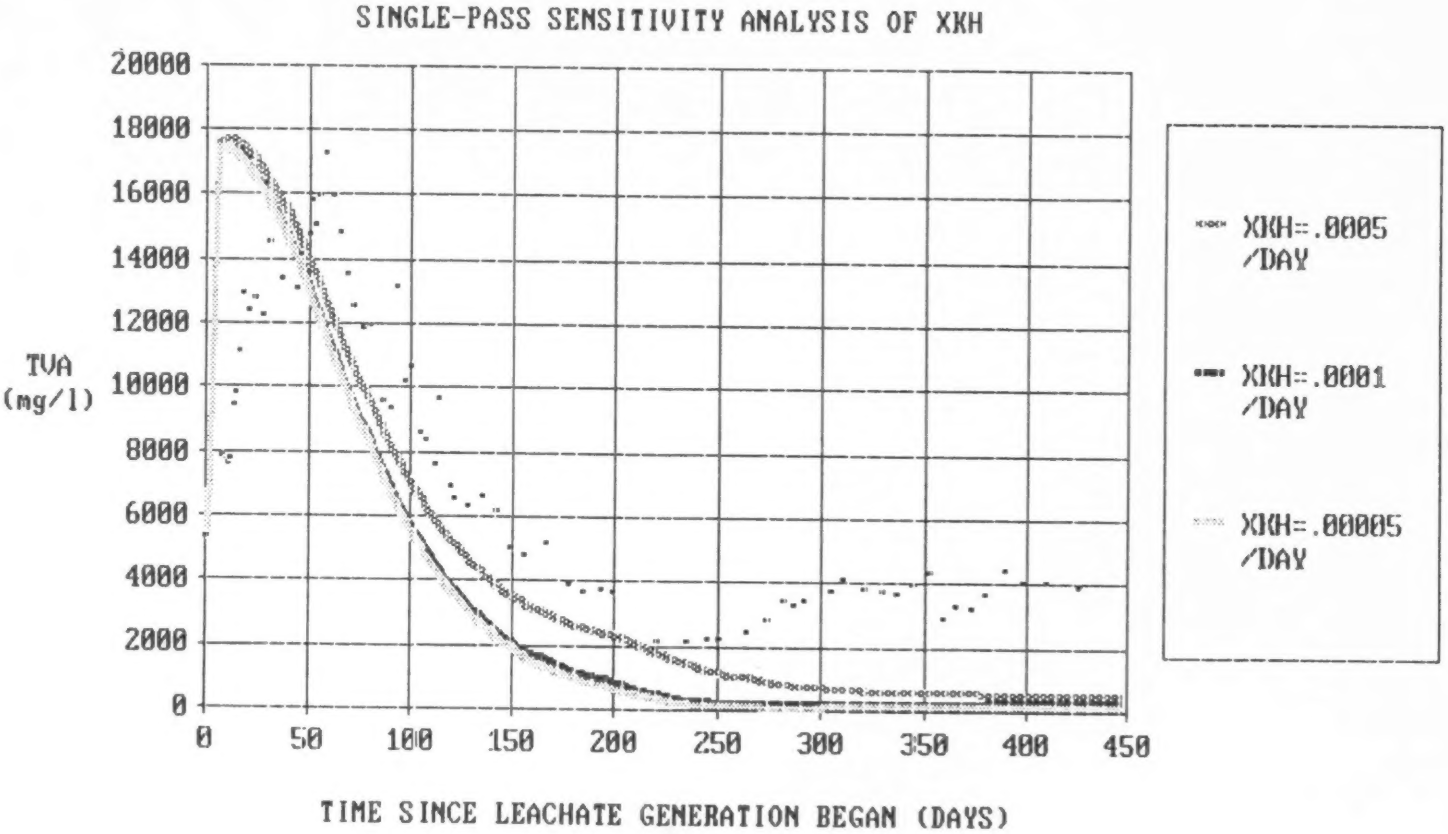
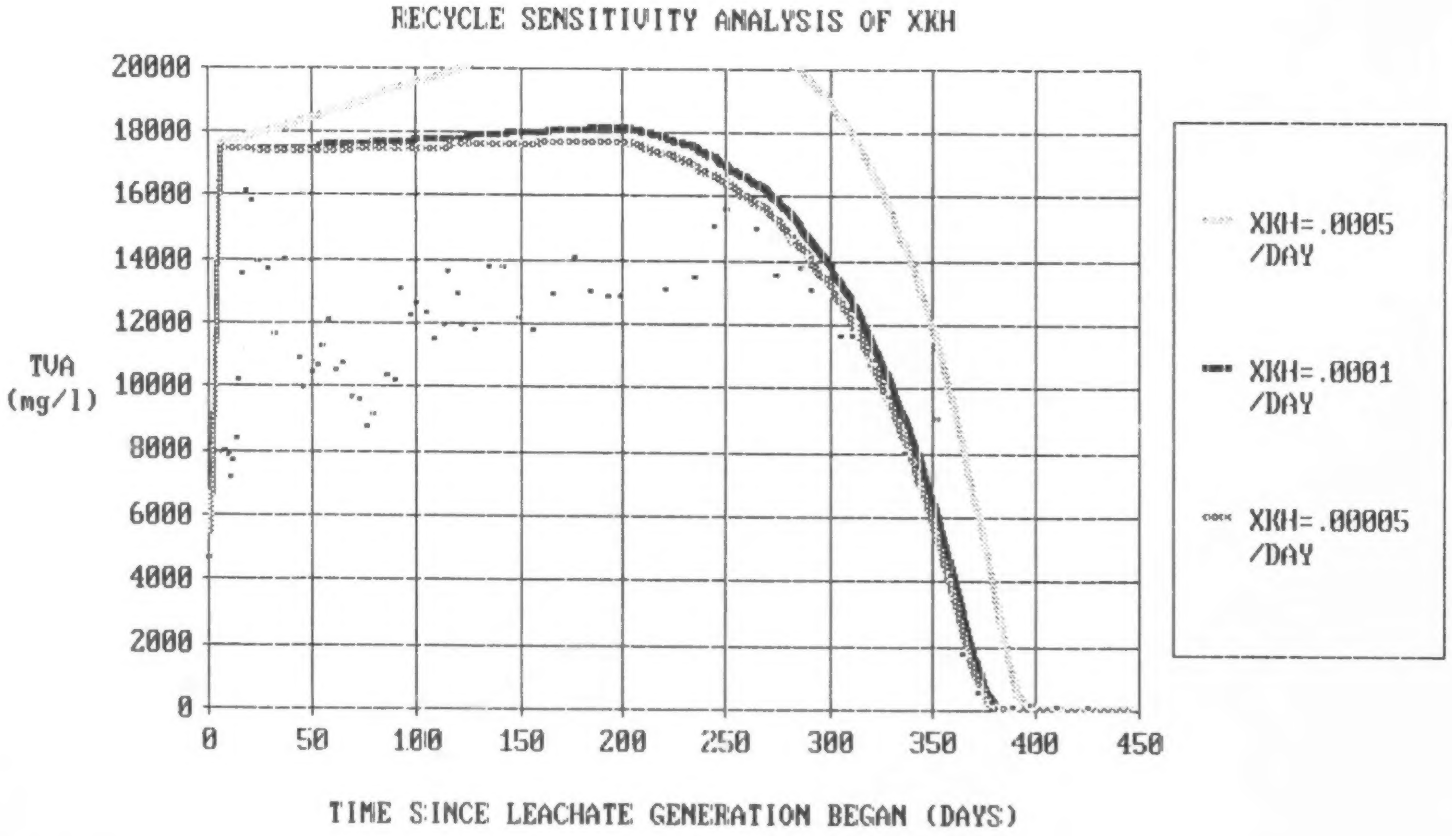

SINGLE-PASS SENSITIVITY ANALYSIS OF XKH
TVA (mg/l)
20000
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
0
50
100
150
200
250
300
350
400
450
TIME SINCE LEACHATE GENERATION BEGAN (DAYS)
XKH=.0005 /DAY
XKH=.0001 /DAY
XKH=.00005 /DAY

FIGURE 29


RECYCLE SENSITIVITY ANALYSIS OF XKH
TVA (mg/l)
20000
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
0
50
100
150
200
250
300
350
400
450
TIME SINCE LEACHATE GENERATION BEGAN (DAYS)
XKH=.0005 /DAY
XKH=.0001 /DAY
XKH=.00005 /DAY

volatile acid concentration with increase in hydrolysis rate constant supports the hypothesis of Eastman and Ferguson (1981) that hydrolysis is the rate-limiting step in the acid phase of anaerobic processes. Thus, increasing the rate of production of hydrolysis products also produced a corresponding increase in volatile acid production.

The effects of changing the maximum rate constant for acidogenesis, Q_A, are examined in Figures 30 and 31. Decreasing the value of Q_A from 3.5/day to 3.0/day increased the magnitude of both single-pass and recycle volatile acid curves. This result can also be explained by a hydrolysis rate-limitation in the acid phase of anaerobic digestion. Increasing the maximum substrate utilization rate constant of acidogenesis alone should have increased the rate of production of volatile acids. However, if the increase in the maximum rate constant for acidogenesis produced a decrease in the concentration of hydrolysis products greater than the increase in the maximum rate constant for acidogenesis, the overall rate of production of volatile acids would decrease according to Equation(59).

Moderate changes in the half-velocity constant for acidogenesis, K_A, from 100 mg/L to 500 mg/L had very little effect on the single-pass and recycle simulations as shown in Figures 32 and 33. Results of adjustments to the yield coefficient for acidogenesis are shown in Figures 34 and 35. Moderate changes in the yield coefficient for acidogens, Y_A, from 0.75 to 1.25 also had virtually no effect on the simulations for either flow regime. Finally, Figures 36 and 37 reveal that adjustments to the value of the decay rate for acidogens had no impact on either the single-pass or recycle simulations. Thus, the major parameter of consequence for acidogenesis was the maximum rate constant.

Adjusting the value of the maximum rate constant for methanogenesis, Q_M, from 1.5/day to 2.5/day, as shown in Figures 38 and 39, had virtually no effect on the single-pass curve largely because it was overshadowed by the effects of washout. A significant impact on the recycle curve was observed adjusting the rate constant for methanogenesis. Decreasing the value of Q_M from 1.9/day to 1.5/day increased the time it took for the volatile acid concentration curve to descend toward zero. Increasing Q_M to 2.5/day shortened that time.

Adjustment of the value of the half-velocity constant for methanogenesis affected the simulations to a limited extend as shown in Figures 40 and 41. Increasing the value of K_M from 100 mg/L to 1000 mg/L increased the time at which the concentration of volatile acids descended toward zero in the recycle mode. The same increase in K_M caused a slight increase in the level at which the volatile acid concentration leveled off in the single-pass mode. Both of these results reflect a decrease in the overall rate of removal of volatile acids by methanogenesis.

As illustrated in Figures 42 and 43, the yield coefficient for methanogens, Y_M, had a very small impact on the single-pass simulation, yet a very dramatic impact on the recycle simulation. A decrease in the yield coefficient from 0.05 to 0.005 prevented the establishment of the population

FIGURE 30

SINGLE-PASS SENSITIVITY ANALYSIS OF QA

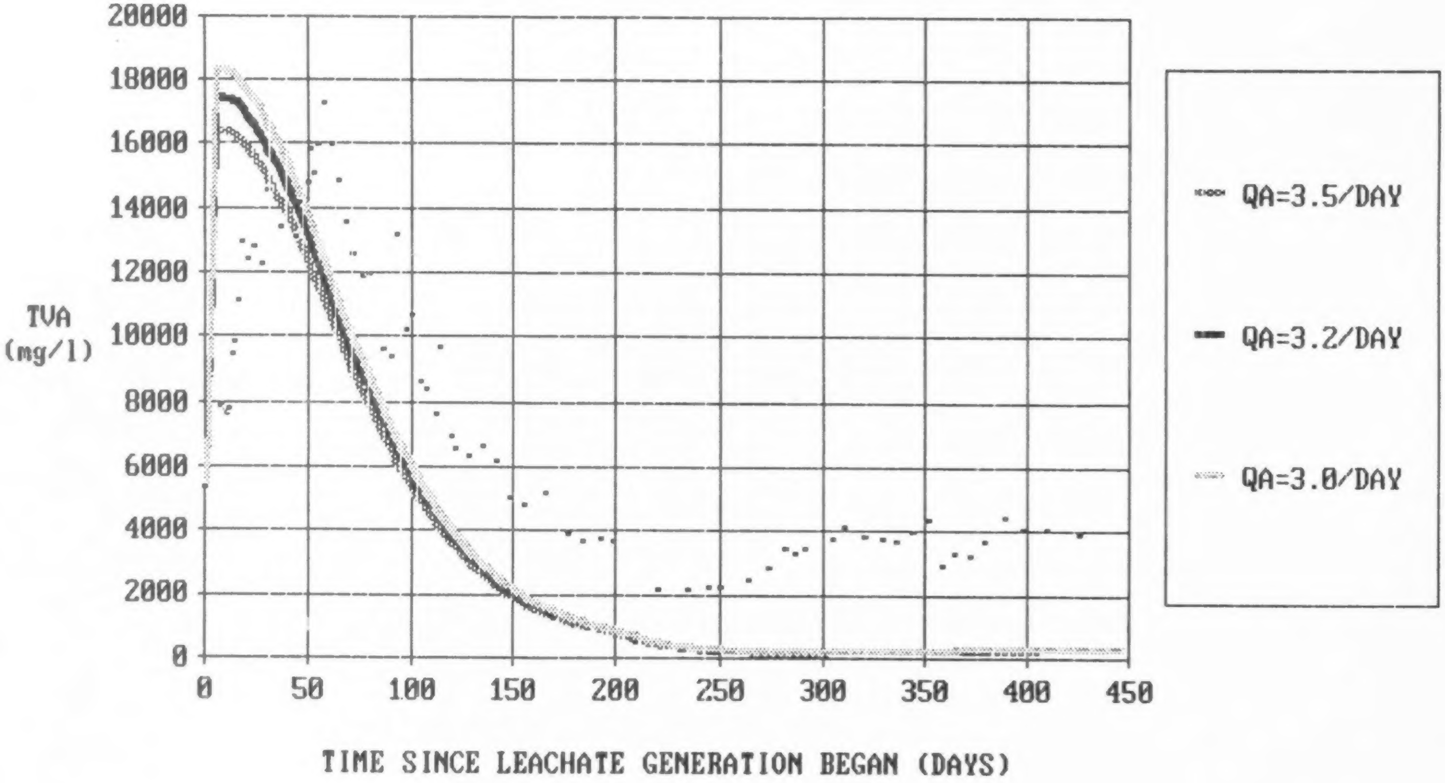


FIGURE 31

RECYCLE SENSITIVITY ANALYSIS OF QA

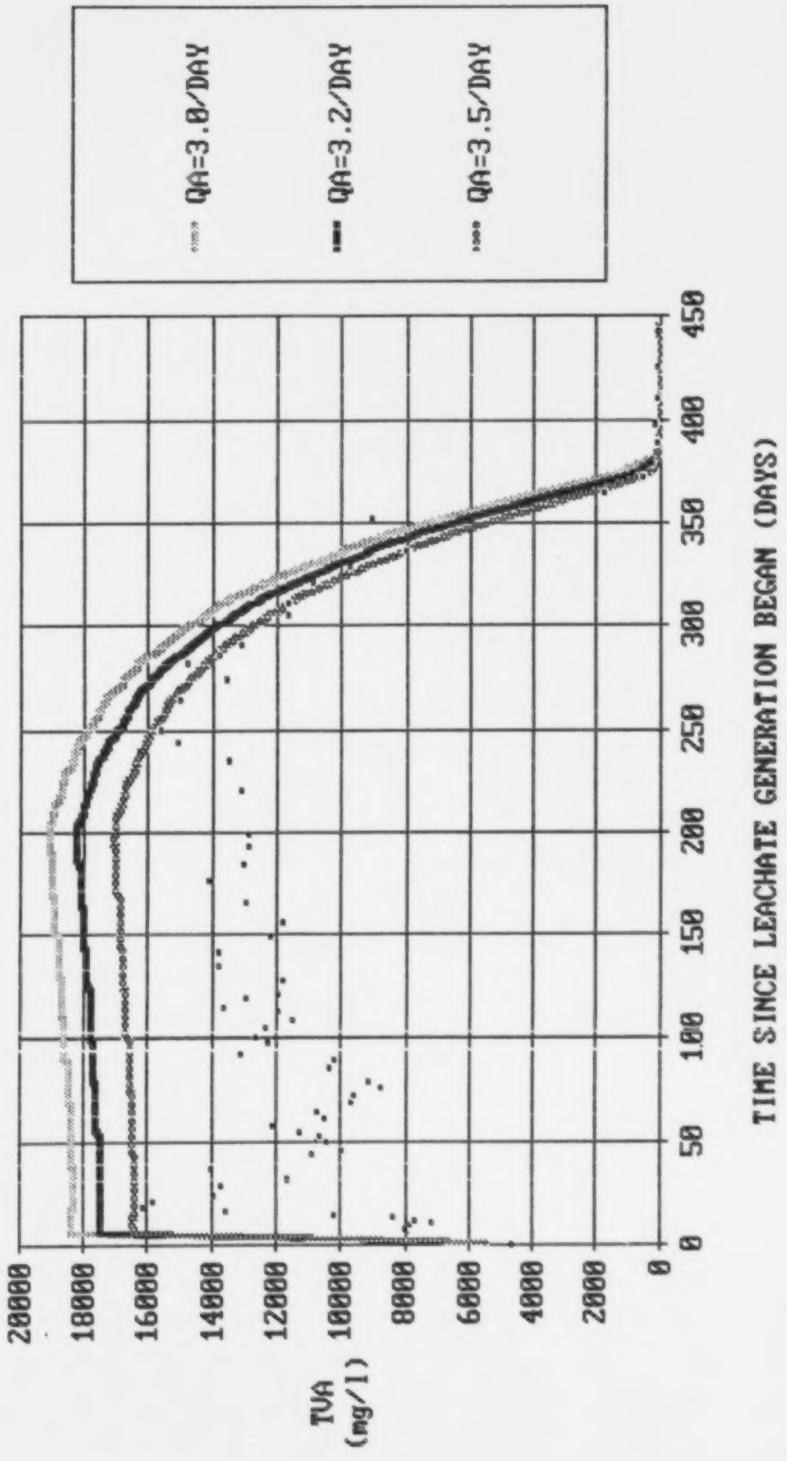


FIGURE 32

SINGLE-PASS SENSITIVITY ANALYSIS OF XKA

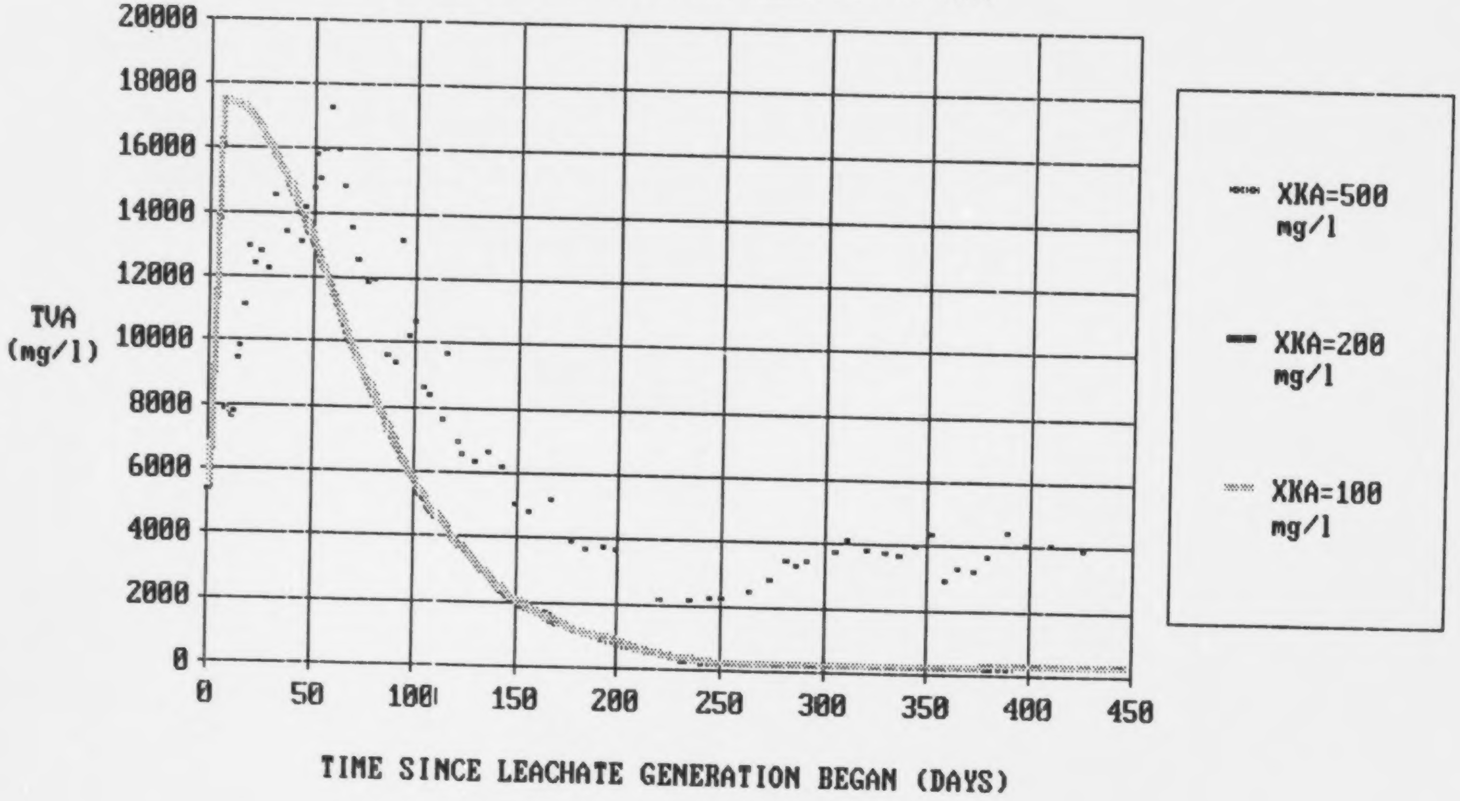


FIGURE 33

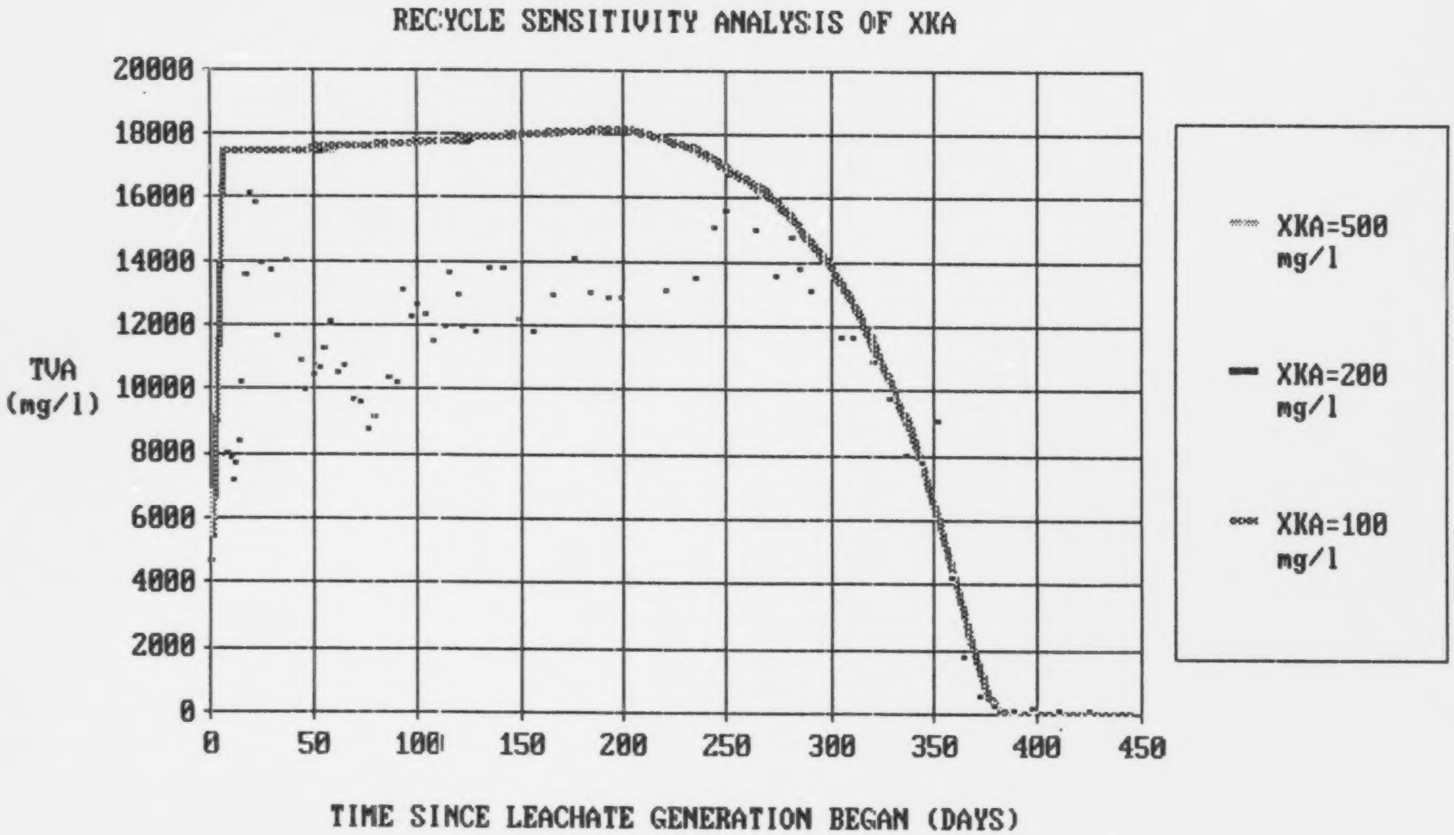

RECYCLE SENSITIVITY ANALYSIS OF XKA
TVA (mg/l)
20000
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
0
50
100
150
200
250
300
350
400
450
TIME SINCE LEACHATE GENERATION BEGAN (DAYS)
XKA=500 mg/l
XKA=200 mg/l
XKA=100 mg/l

FIGURE 34

SINGLE-PASS SENSITIVITY ANALYSIS OF YA

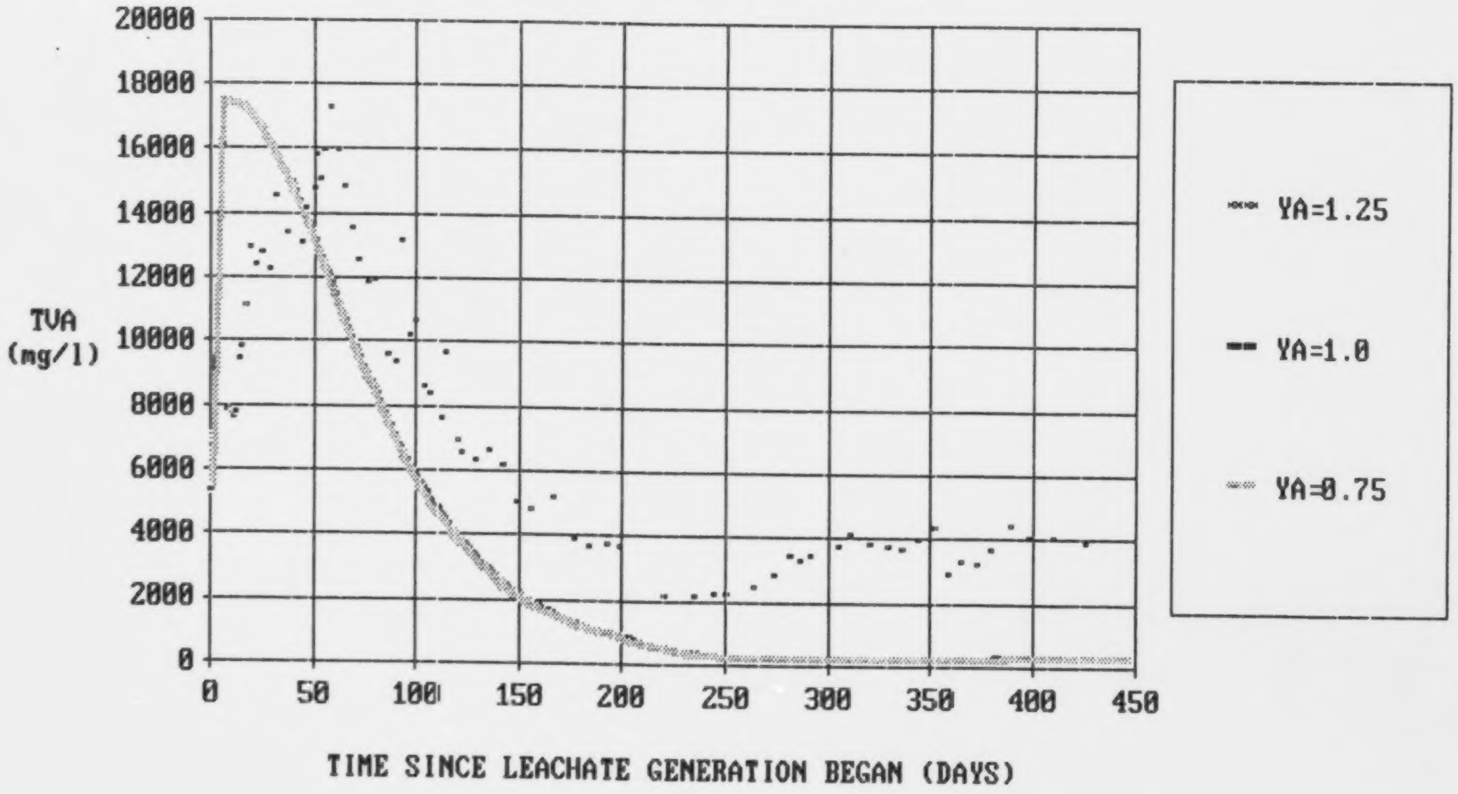


FIGURE 35

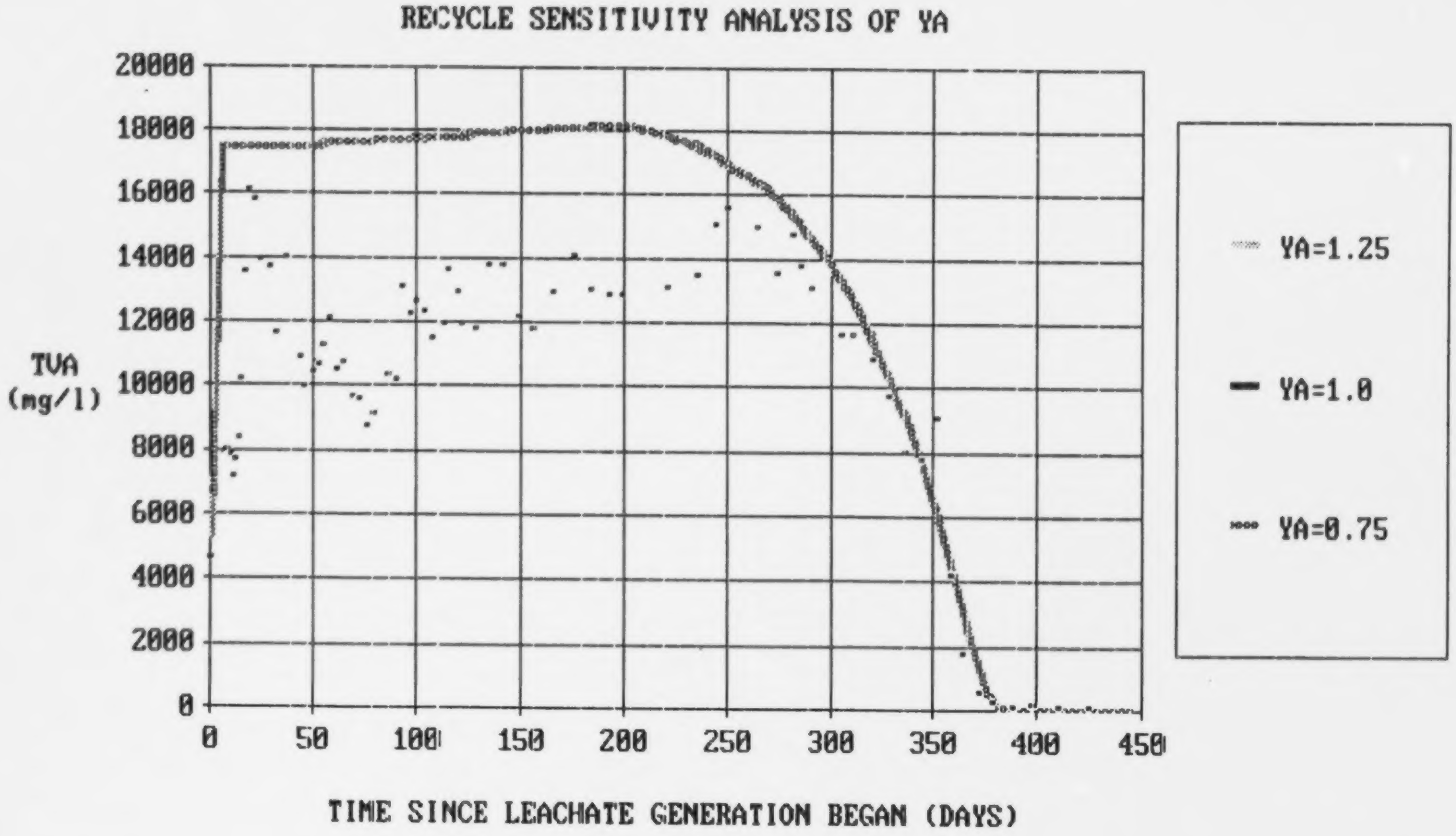

RECYCLE SENSITIVITY ANALYSIS OF YA
TVA (mg/l)
20000
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
0
50
100
150
200
250
300
350
400
450
TIME SINCE LEACHATE GENERATION BEGAN (DAYS)
YA=1.25
YA=1.0
YA=0.75

FIGURE 36

SINGLE-PASS SENSITIVITY ANALYSIS OF XKDA

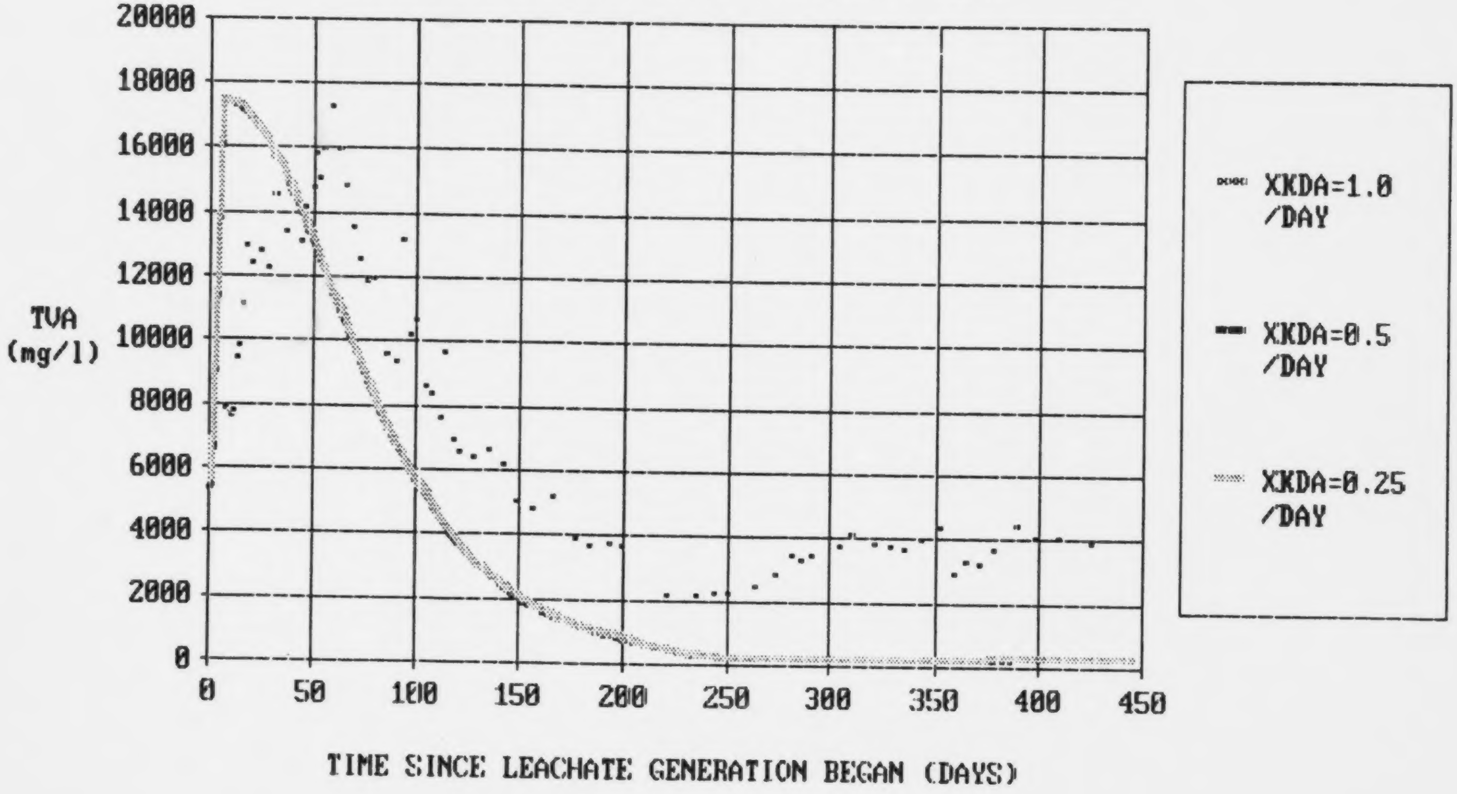


FIGURE 37

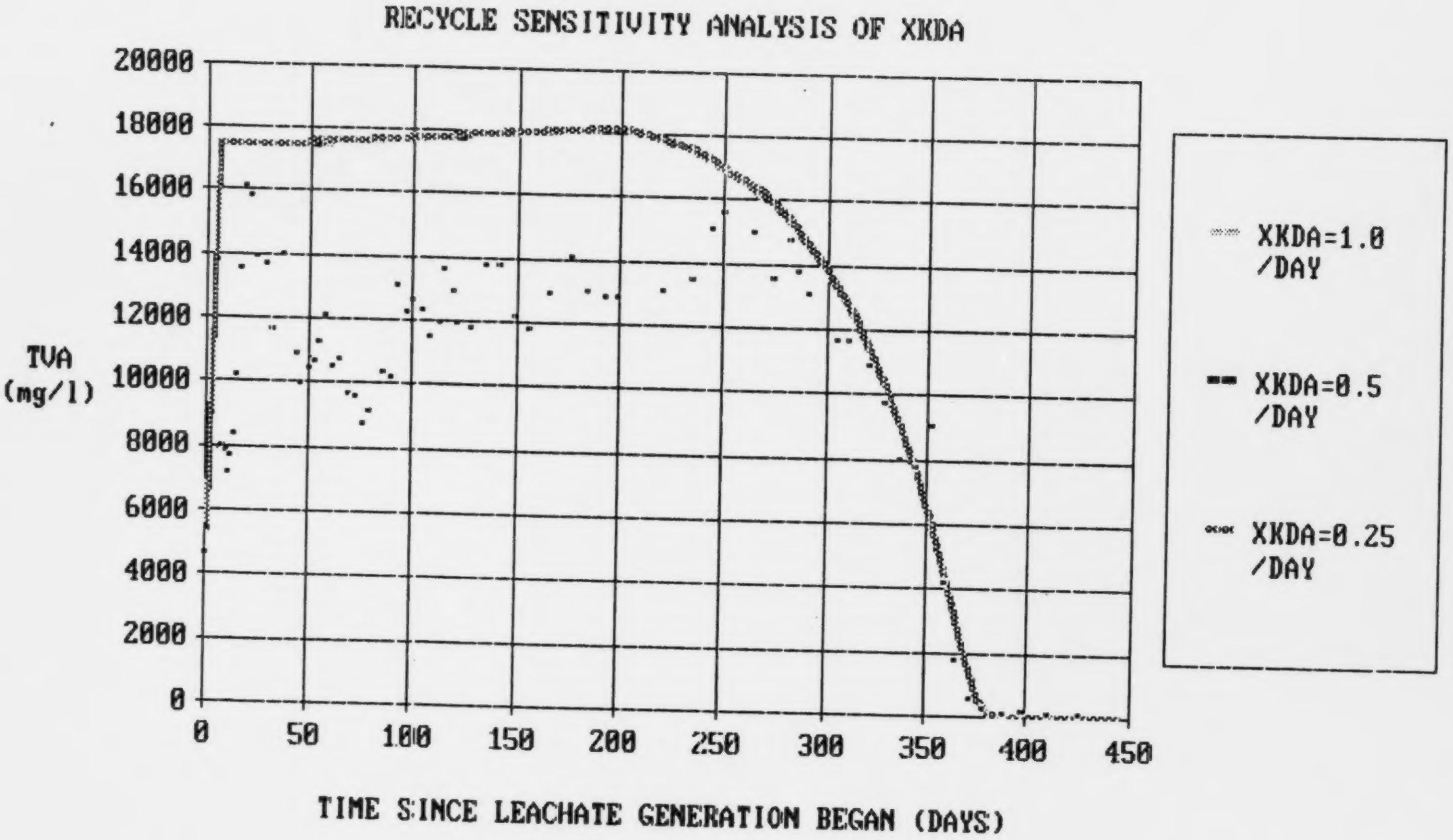

RECYCLE SENSITIVITY ANALYSIS OF XKDA
TVA (mg/l)
20000
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
0
50
100
150
200
250
300
350
400
450
TIME SINCE LEACHATE GENERATION BEGAN (DAYS)
XKDA=1.0 /DAY
XKDA=0.5 /DAY
XKDA=0.25 /DAY

FIGURE 38

SINGLE-PASS SENSITIVITY ANALYSIS OF QM

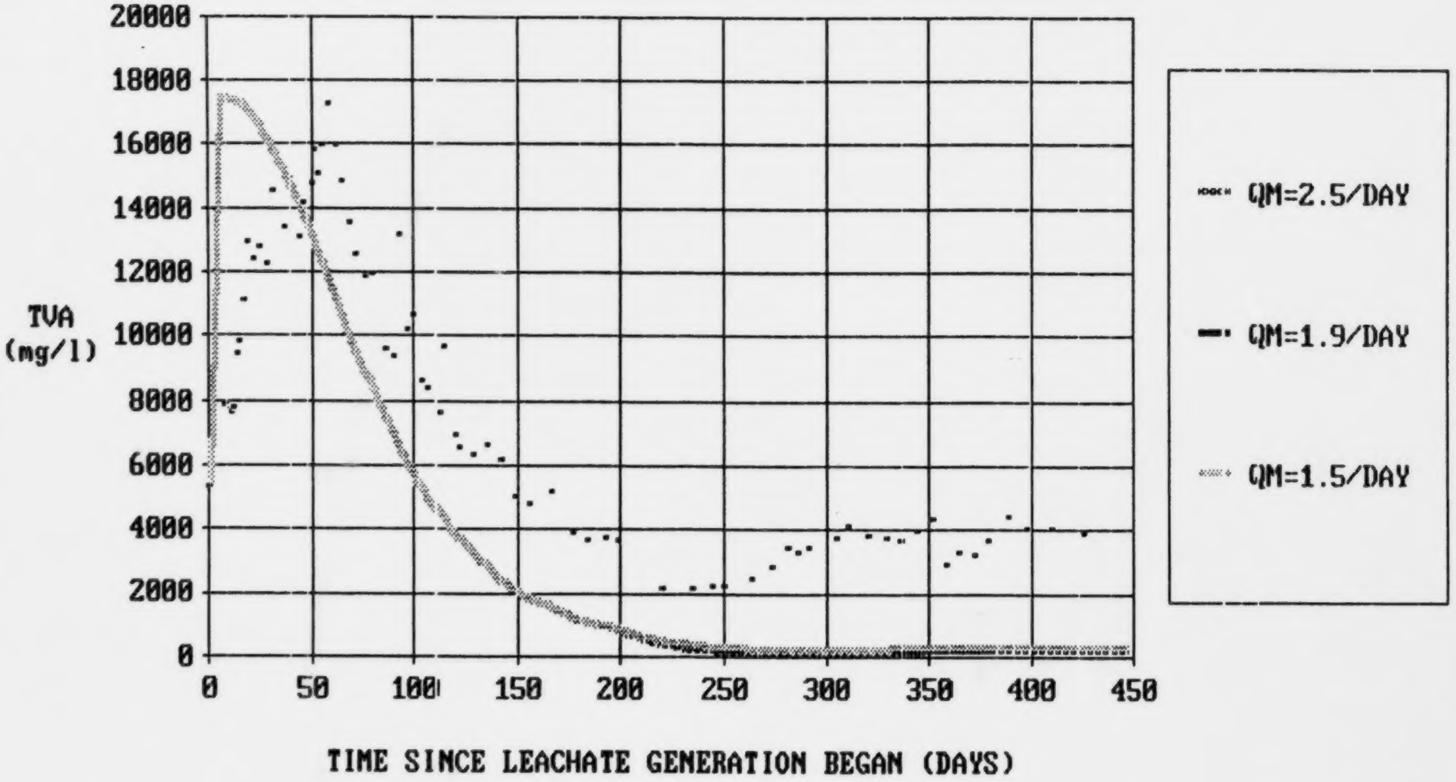


FIGURE 39

RECYCLE SENSITIVITY ANALYSIS OF QM

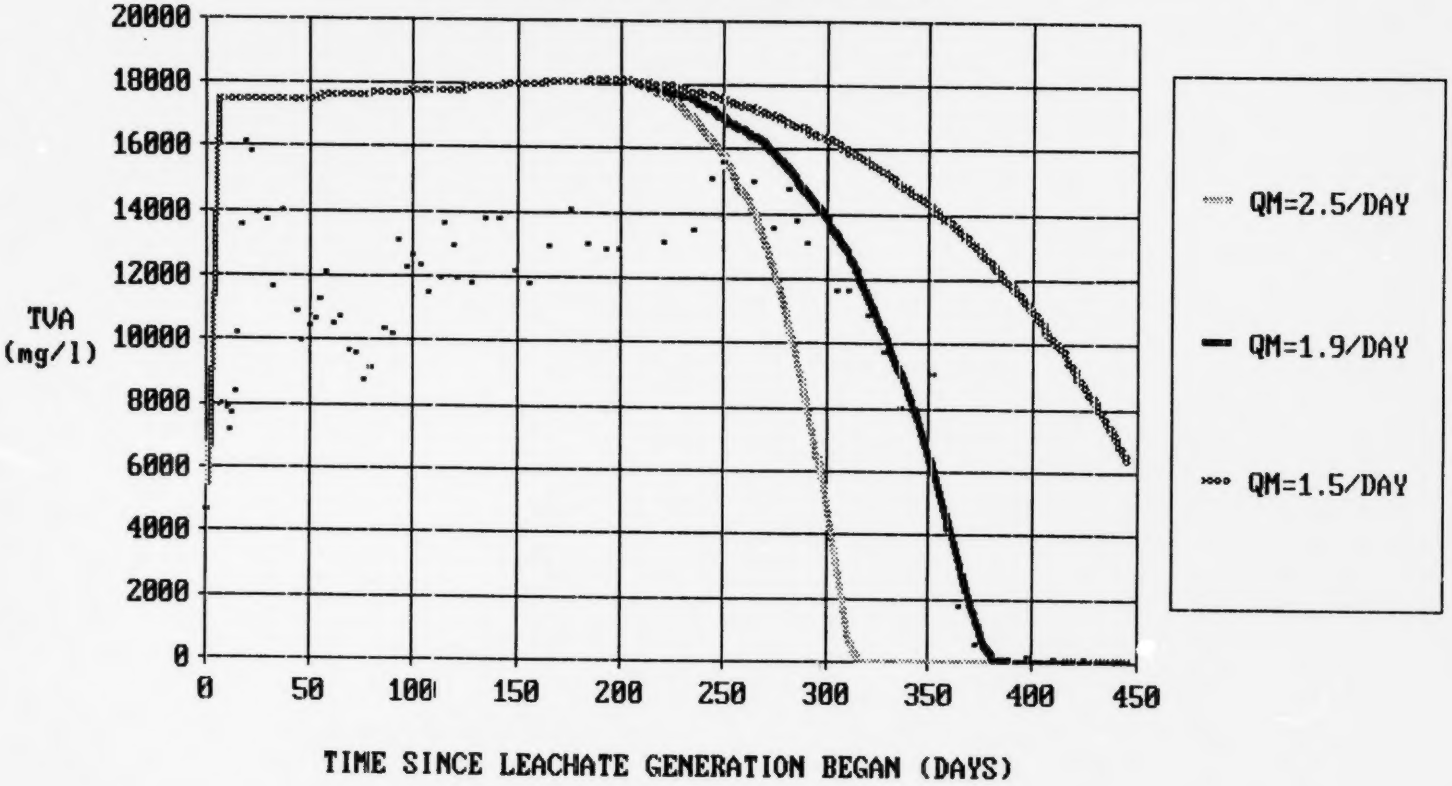


FIGURE 40

SINGLE-PASS SENSITIVITY ANALYSIS OF XKM

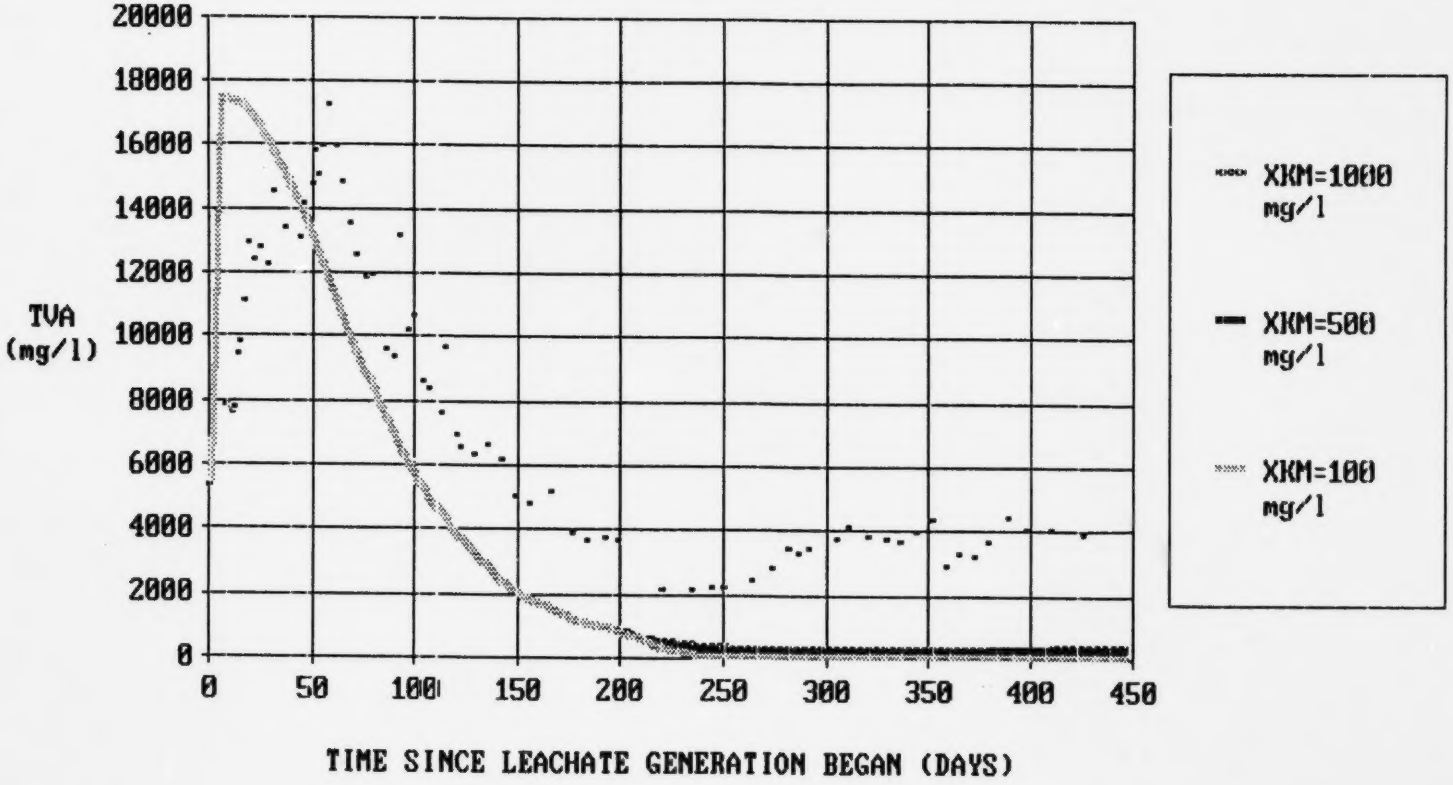


FIGURE 41

RECYCLE SENSITIVITY ANALYSIS OF XKM

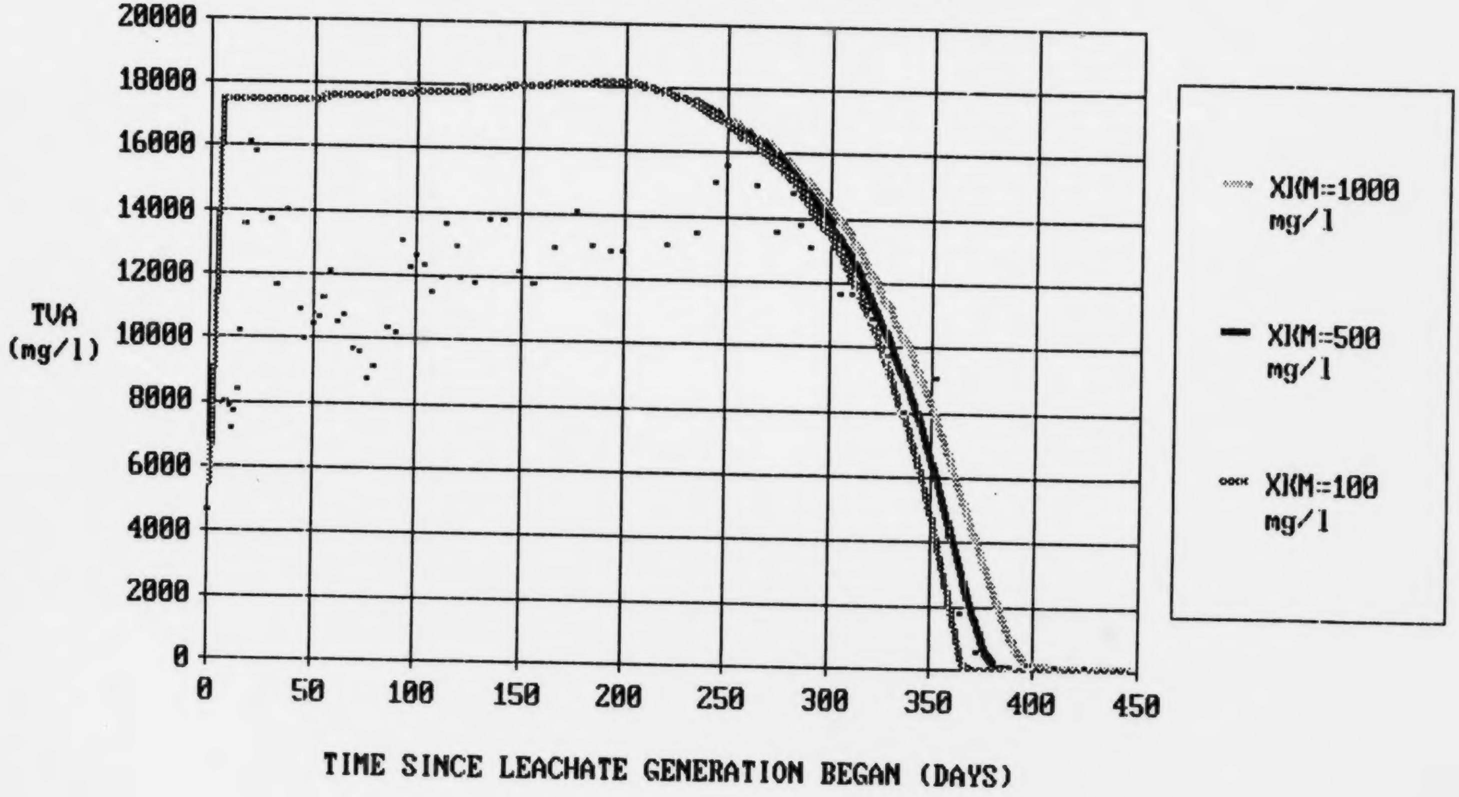


FIGURE 42

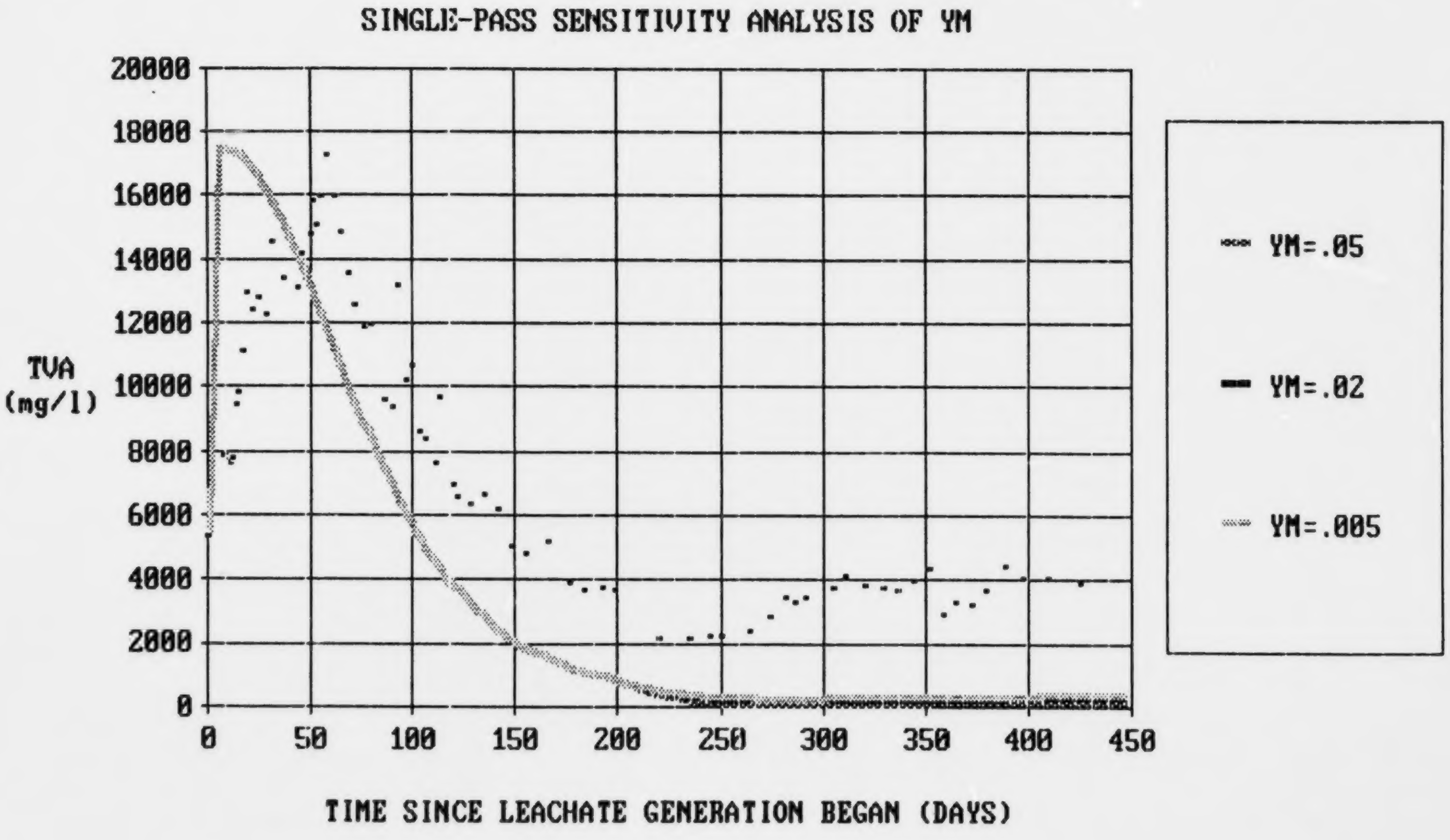
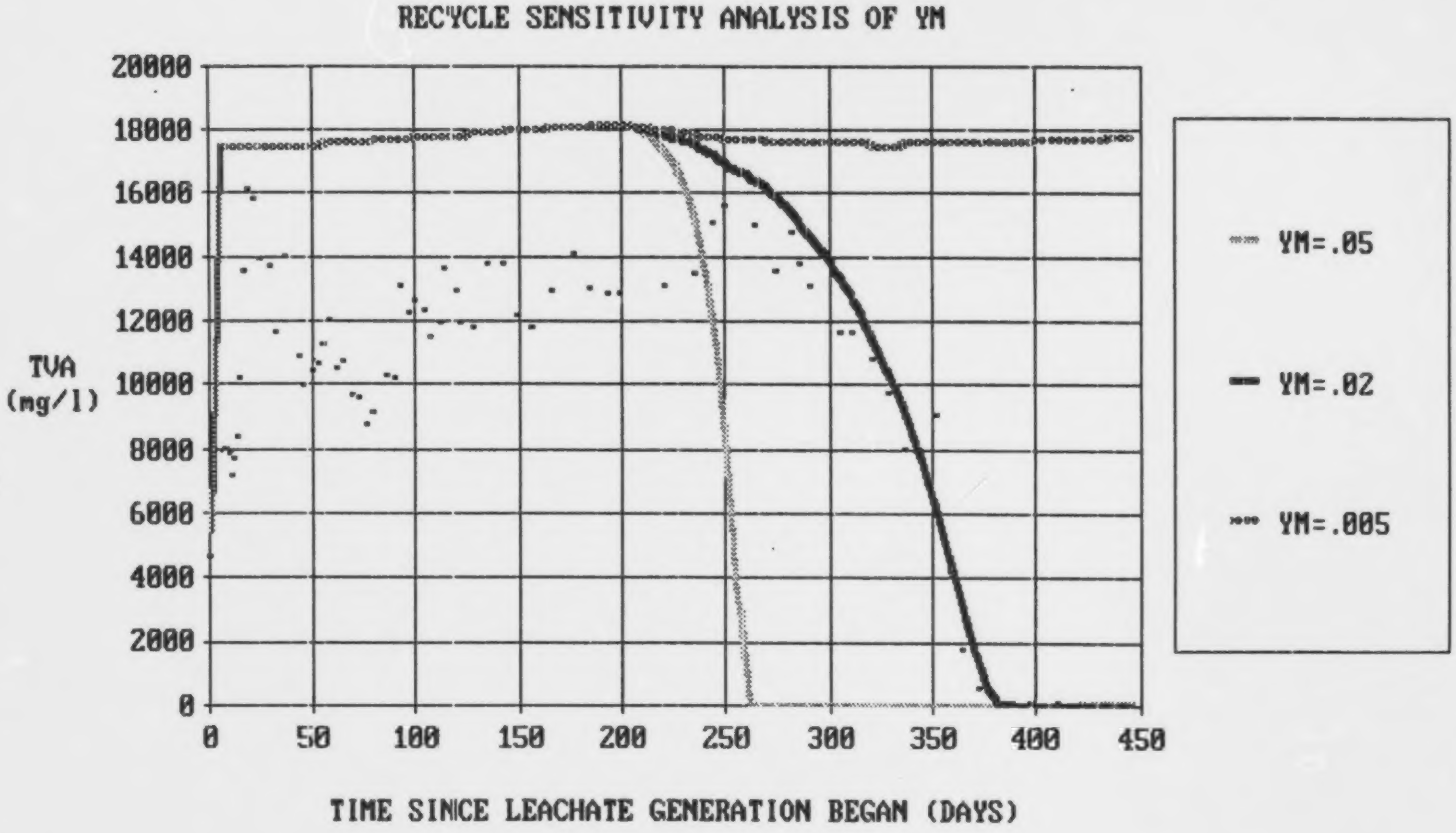

SINGLE-PASS SENSITIVITY ANALYSIS OF YM
TVA (mg/l)
20000
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
0
50
100
150
200
250
300
350
400
450
TIME SINCE LEACHATE GENERATION BEGAN (DAYS)
YM=.05
YM=.02
YM=.005

FIGURE 43


RECYCLE SENSITIVITY ANALYSIS OF YM
TVA (mg/l)
20000
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
0
50
100
150
200
250
300
350
400
450
TIME SINCE LEACHATE GENERATION BEGAN (DAYS)
YM=.05
YM=.02
YM=.005

of methanogens, thereby shortening the time before removal of volatile acids was essentially complete.

The effect of modifications in the decay rate for methanogens on the fit of the simulations to the data is demonstrated in Figures 44 and 45. As observed for changes in the yield coefficient, changes in the decay rate for methanogens from 0.02/day to 0.05/day produced an effect similar to decreasing the yield coefficient for methanogens, resulting in a flat volatile acid curve. Correspondingly, a decrease in the decay rate, K_{DM}, shortened the time for essentially complete removal of the volatile acids.

A summary of the results of the sensitivity analysis is presented in Table 3. One striking observation derived from the results of the sensitivity analysis was that the establishment of methanogenesis had very little impact on the volatile acid profile for the single-pass system, while it had a very important impact on the volatile acid profile for the recycle system. Upon closer examination it seems that, in general, the volatile acid profile for the single-pass system was less sensitive to changes in all of the parameters when compared to the response of the recycle system. The most obvious explanation for this behavior was that the washout effects of the single-pass system masked the response of the system to changes in the parameters. Thus it would be expected that a decrease in the rate of moisture application to the single-pass system should cause the single-pass system to behave in a manner more similar to the recycle system as the effects of washout were diminished. In fact, examination of an additional set of experiments conducted by Chang (1982) reveals this to be the case as is demonstrated subsequently.

SEPARATE FITTING

Based on further understanding of the model gained from the simultaneous fitting exercise and sensitivity analysis, an improved fit of the model to the data from the Schaffer (1986) and Yari (1986) experiment was attempted. In this case, the two leachate management options were fit separately, i.e., a different set of kinetic constants was fit for each of the two leachate management options. Additionally, the fit of the methane production simulation to experimental data was examined to provide an additional test of GTLEACH-I.

Kinetic parameters selected in the final separate fit for the single-pass simulation are:

Hydrolysis	Acidogenesis	Methanogenesis
K_H = 0.0008/day	Q_A = 3.2/day	Q_M = 1.5/day
	K_A = 200 mg/L	K_M = 500 mg/L
	K_{DA} = 0.5/day	K_{DM} = 0.02/day
	Y_A = 1.0	Y_M = 0.028
		Methanogen lay = 200 days

FIGURE 44

SINGLE-PASS SENSITIVITY ANALYSIS OF XKDM

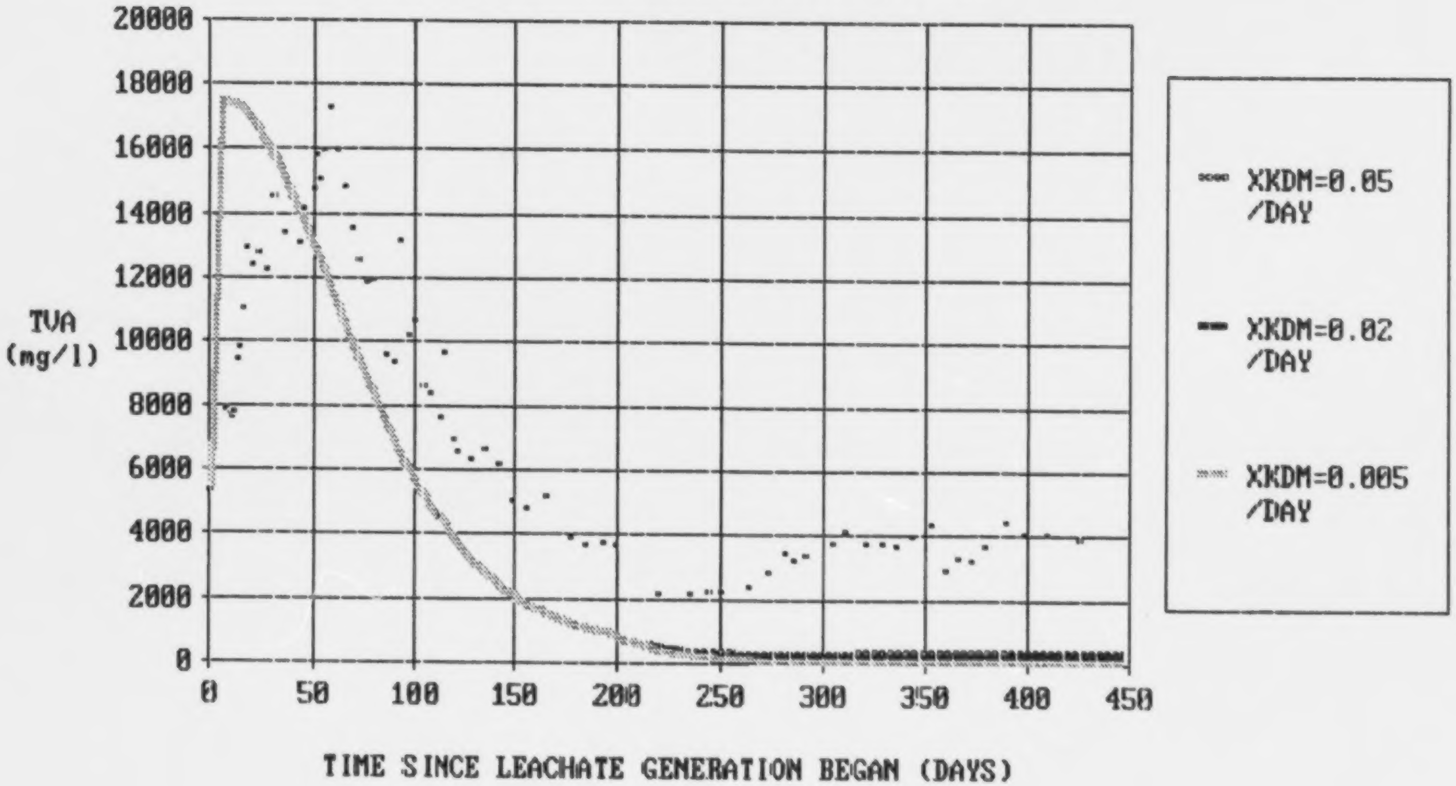


FIGURE 45

RECYCLE SENSITIVITY ANALYSIS OF XKDM

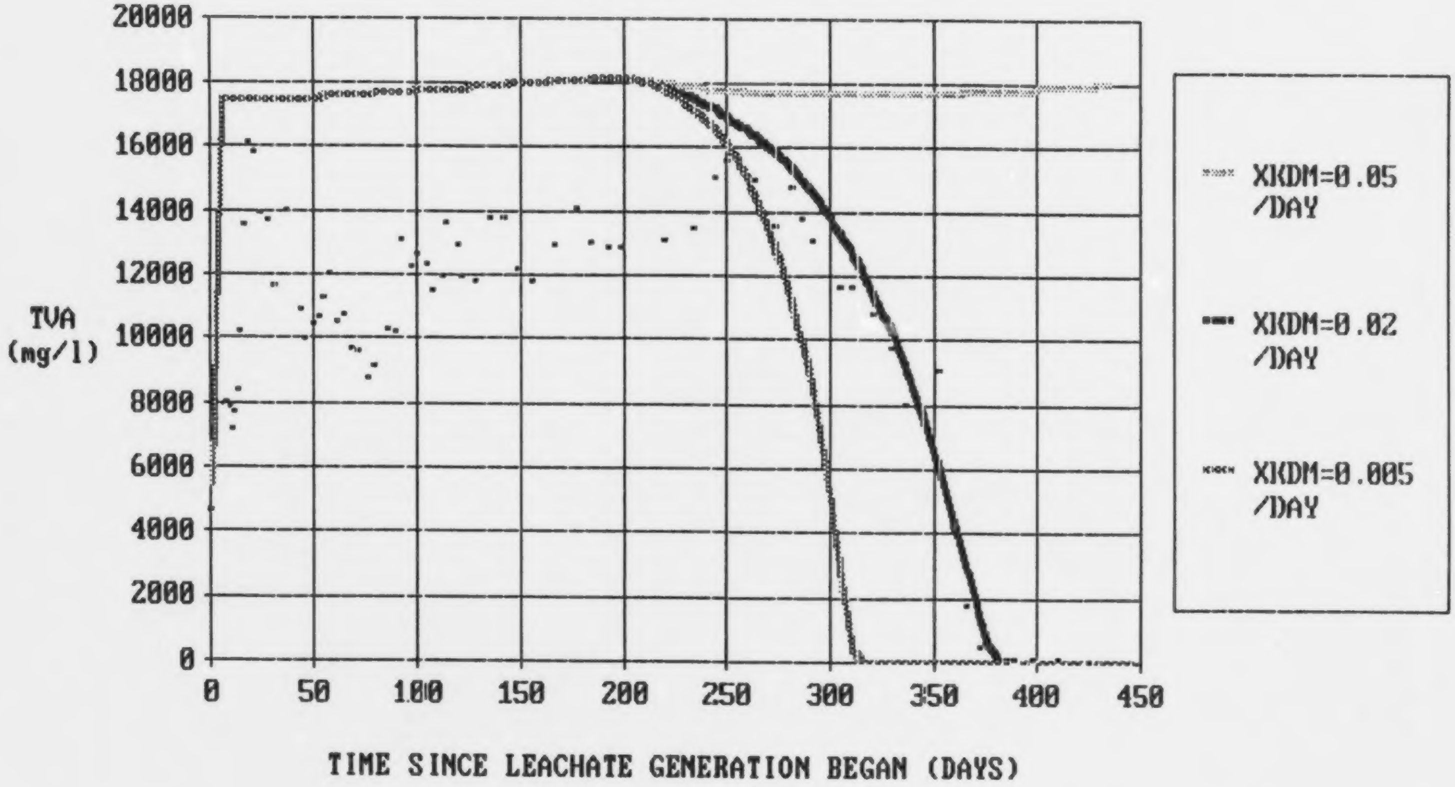


TABLE 3

RESULTS OF SENSITIVITY ANALYSIS ON VOLATILE ACID CONCENTRATION PROFILE

	Single-pass Mode	Recycle Mode
Increase # CSTR's	approach plug flow	little or no effect
Increase K_H	increase magnitude of concentration profile	increase magnitude of concentration profile
Decrease Q_A	increase magnitude of concentration profile	increase magnitude of concentration profile
Changes in K_A	little or no effect	little or no effect
Changes in Y_A	little or no effect	little or no effect
Changes in K_{sA}	little or no effect	little or no effect
Decrease Q_M	little or no effect	increase time to removal
Increase K_M	increase final concentration attained	increase time to removal by methanogens
Decrease Y_M	little or no effect	increase time to removal by methanogens
Decrease K_{sM}	little or no effect	increase time to removal by methanogens

The changes which occurred in the kinetic parameters selected in the separate fitting from those in the simultaneous fitting of the single-pass data were: the hydrolysis rate constant increased from 0.0001 to 0.0008/day, the maximum substrate utilization rate for methanogenesis decreased from 1.9 to 1.5/day, and the yield coefficient for methanogenesis increased from 0.02 to 0.028. In addition to changes in kinetic parameters, the initial concentration of methanogens was decreased from 10 mg/L to 0.1 mg/L. Figures 46 and 47 compare the results obtained from the separately fit parameters with those from the simultaneous fit for the single-pass data.

A markedly improved fit of the volatile acid data was obtained in the separately fit single-pass simulation when compared to the simultaneous fit. The separately fit simulated volatile acid concentration curve closely followed the experimental data as it decreased from the initial peak. The experimental concentration of volatile acids reached a low point near Day 225 then increased slightly for a period of 75 days until leveling out. The separately fit simulation did not follow these latter trends exactly, but rather smoothed them by slightly overestimating the low and underestimating the eventual leveling out of the curve. This was still a substantial improvement over the simultaneously fit simulation.

Unfortunately, improvement in the simulation of methane production could not be obtained without sacrificing the fit of the volatile acid data. In order to elevate the levels of methane production in accordance with the data, the maximum substrate utilization rate or the yield coefficient for methanogens would have been increased (effectively increasing the growth rate of methanogens). The resulting increase in methanogenesis would have decreased the concentration of volatile acids in the leachate resulting in a poorer fit of the volatile acid data.

Figures 48 and 49 compare two alternative improved fits to the recycle data of Schaffer (1986) and Yari (1986). The two sets of kinetic parameters employed in these fits were:

SEPARATE FIT 1

Hydrolysis	Acidogenesis	Methanogenesis
K_H = 0.0005/day	Q_A = 3.5/day	Q_M = 1.9/day
	K_A = 200 mg/L	K_M = 500 mg/L
	K_{DA} = 1.0/day	K_{DM} = 0.02/day
	Y_A = 0.5	Y_M = 0.03

FIGURE 46

COMPARISON OF SIMULATED VOLATILE ACID CONCENTRATION
WITH EXPERIMENTAL DATA (SCHAFFER 1986, YARI 1986), SINGLE-PASS

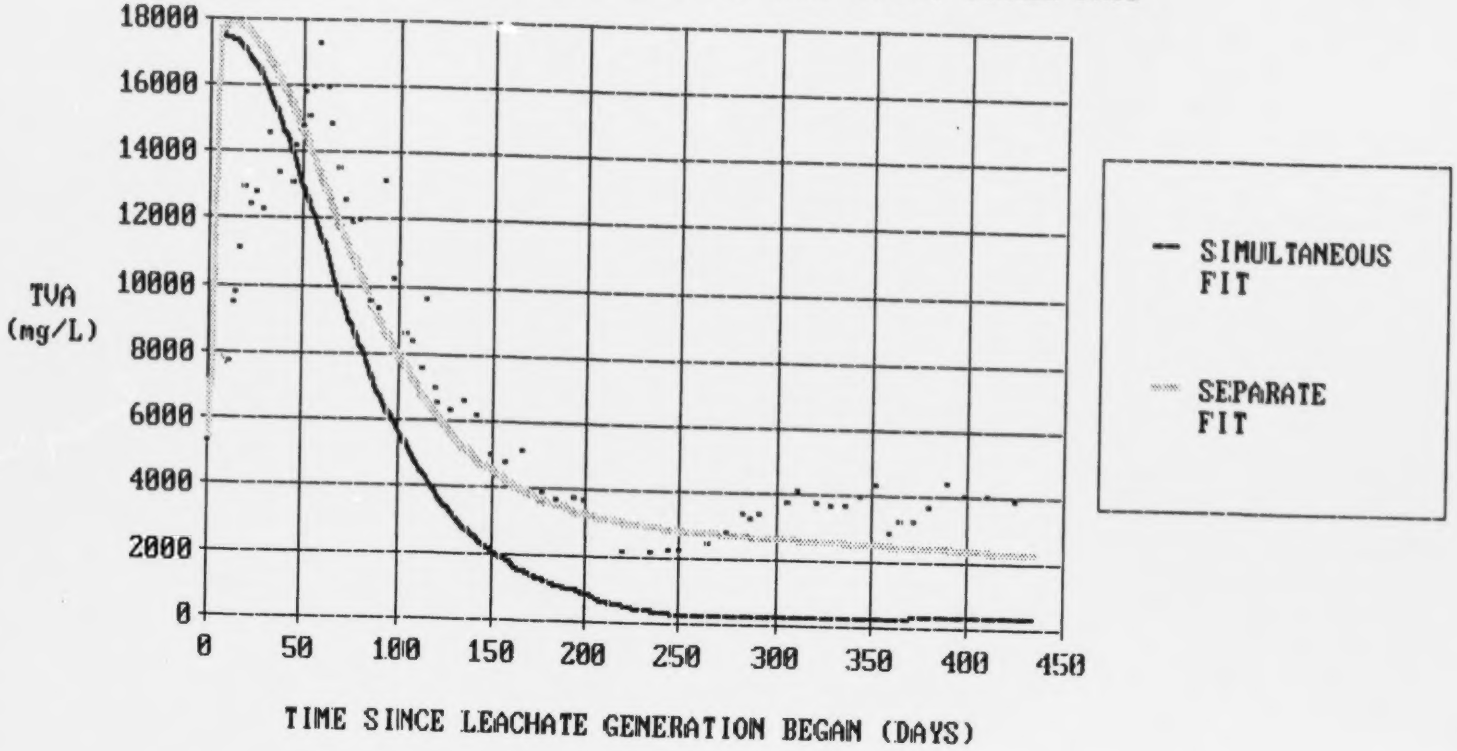


FIGURE 47

COMPARISON OF SIMULATED METHANE PRODUCTION
WITH EXPERIMENTAL DATA (SCHAFFER 1986, YARI 1986), SINGLE-PASS

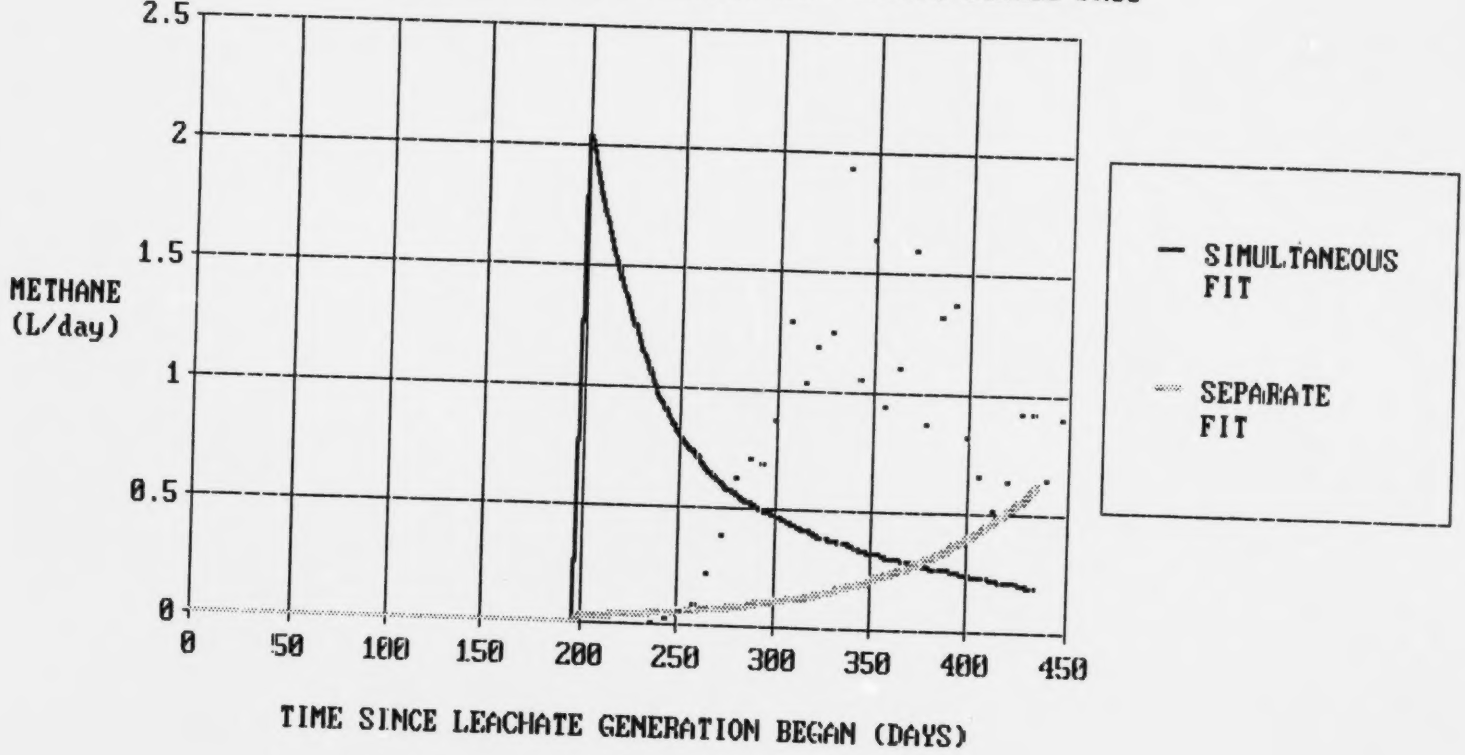


FIGURE 48

COMPARISON OF SIMULATED VOLATILE ACID CONCENTRATION WITH EXPERIMENTAL DATA (SCHAFFER 1986, YARI 1986), RECYCLE

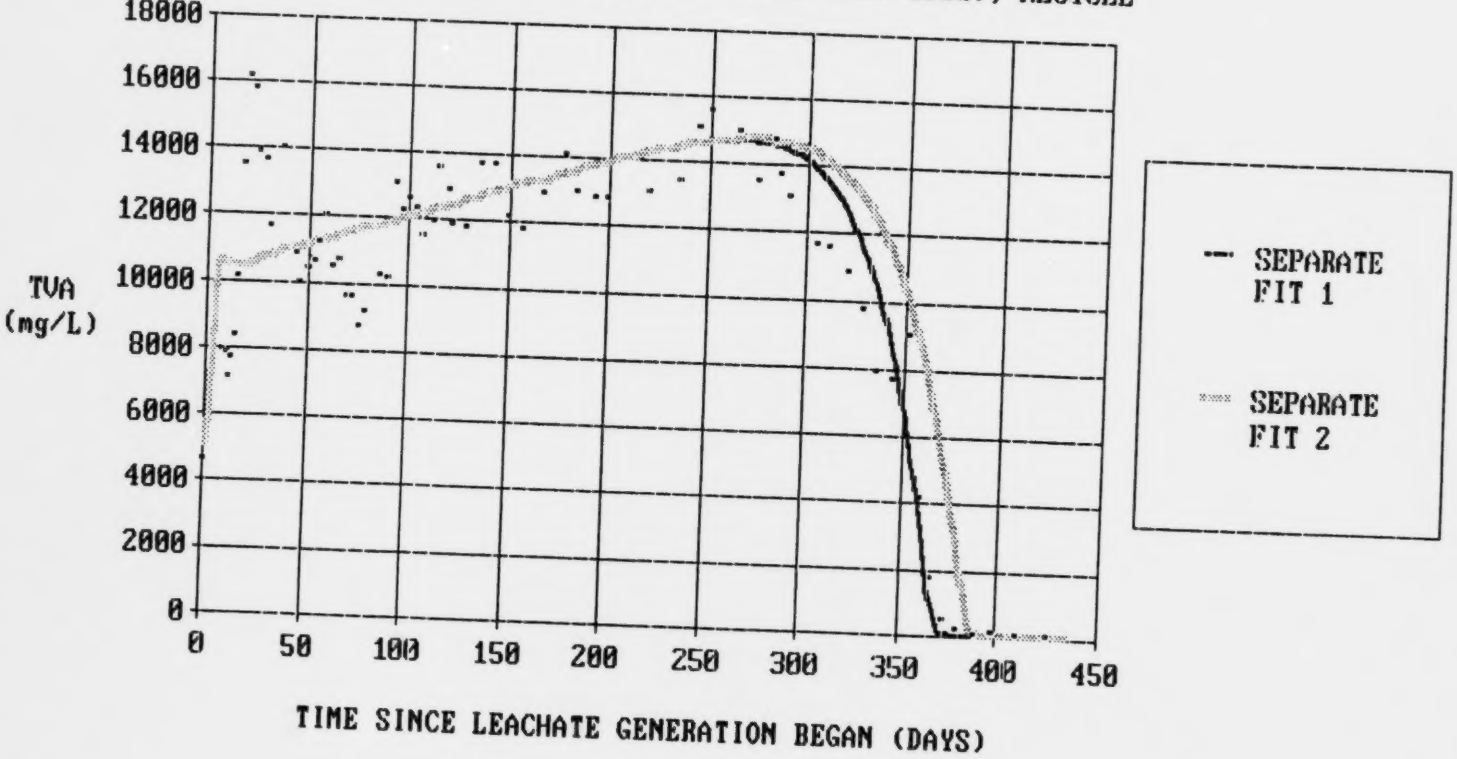
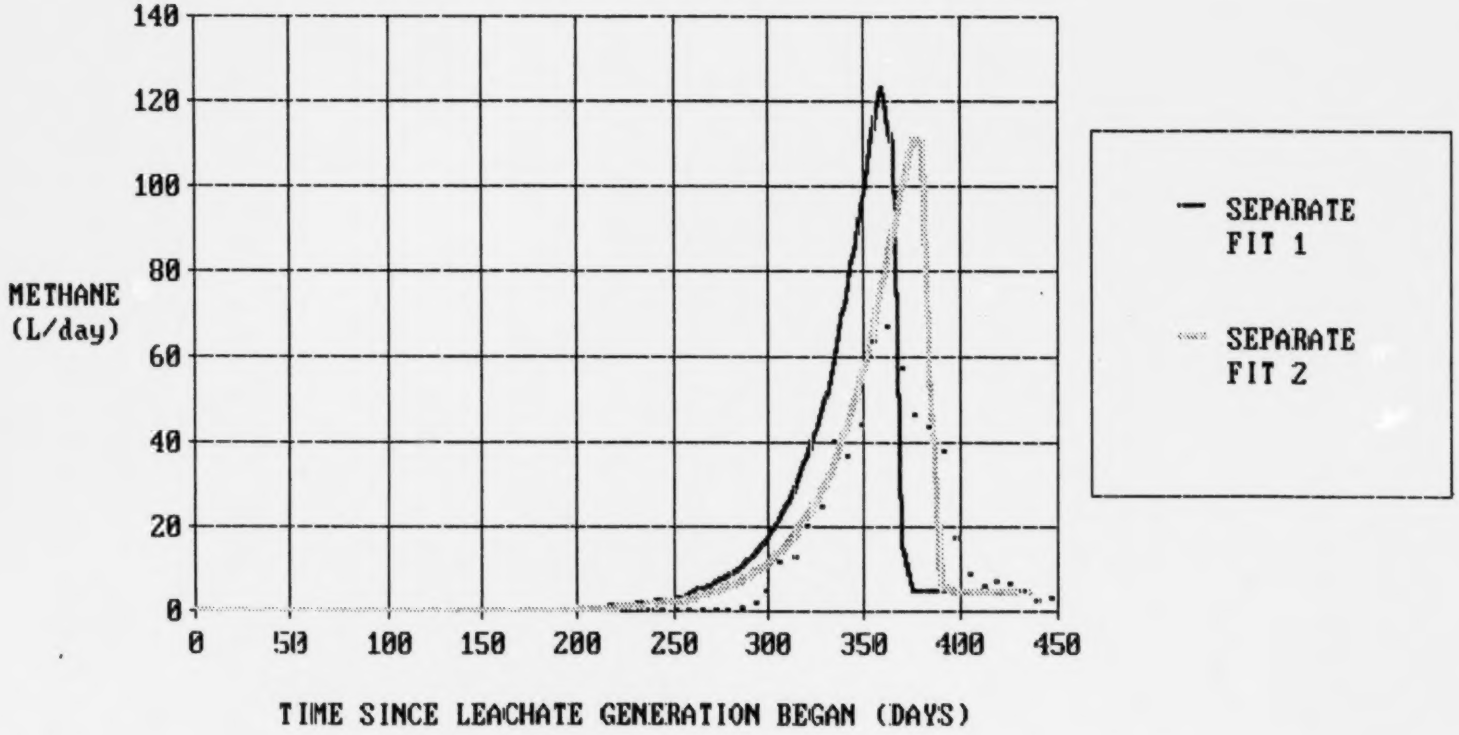


FIGURE 49

COMPARISON OF SIMULATED METHANE PRODUCTION
WITH EXPERIMENTAL DATA (SCHAFFER 1986, YARI 1986), RECYCLE



SEPARATE FIT 2

Hydrolysis	Acidogenesis	Methanogenesis
K_H = 0.0005/day	Q_A = 3.5/day	Q_M = 1.9/day
	K_A = 200 mg/L	K_M = 500 mg/L
	K_{DA} = 1.0/day	K_{DM} = 0.02/day
	Y_A = 0.5	Y_M = 0.028

The only difference between these two sets of parameters was in the yield coefficient for methanogens which was slightly higher for SEPARATE FIT 1 than for SEPARATE FIT 2. Changes in parameters when compared to simultaneously fit parameters were:

- hydrolysis rate constant was increased from 0.0001 to 0.0005/day;
- maximum substrate utilization rate for acidogens was increased from 3.2 to 3.5/day;
- the decay rate for acidogens was increased from 0.5 to 1.0/day;
- the yield coefficient for acidogens was decreased from 1.0 to 0.5;
- the yield coefficient for methanogens was increased from 0.02 to 0.028 and 0.03;
- the initial concentration of hydrolysis products was decreased from 40,000 mg/L to 20,000 mg/L; and
- the initial concentration of methanogens was decreased from 10 mg/L to 1.0 mg/L for both separate fits.

Substantial improvements were effected by the separate recycle fits (Figure 48) when compared to the simultaneous fit (Figure 21). First, decreasing the initial concentration of hydrolysis products decreased the initial peak in volatile acid concentration which, while underestimating the initial peak in the experimental data, allowed the simulation to follow the experimental data closely as the concentration of volatile acids gradually increased from Day 50 to Day 250. Examination of Figures 48 and 49 indicated that while the higher yield coefficient gave better fit to the volatile acid data as the concentration decreased from Day 275 to Day 400, there was sacrifice in the fit of the methane production data when compared with SEPARATE FIT 2. Decreasing the initial concentration of methanogens in the separate fit served to eliminate the abrupt initiation of methanogenesis observed in the simultaneously fit data (Figure 19). It should be noted, however, that both separate simulations overestimated the peak methane production data.

The key differences between the input parameters of the separately fit single-pass and recycle simulations were:

- initial concentration of hydrolysis products for the single-pass simulation was 40,000 mg/L, and was 20,000 mg/L for the recycle simulation;
- the hydrolysis rate constant was nearly twice as high for the single-pass simulation than for the recycle simulation as observed through the product of the yield coefficient and the maximum substrate utilization coefficient;
- the acidogen growth rate was nearly two times higher for the single-pass simulation than for the recycle simulation as observed through the product of the yield coefficient and the maximum substrate utilization coefficents;
- the decay rate for acidogens was two times higher in the recycle simulation than in the single-pass simulation; and,
- the maximum substrate utilization rate for methanogens was lower in the single-pass simulation than in the recycle simulation.

The higher initial concentration of hydrolysis products in the single-pass simulation when compared to the recycle simulation can be attributed to the greater volume of moisture circulating through the recycle system when compared to the single-pass system. This would act initially to dilute the hydrolysis product concentration to a greater degree in the recycle system than in the single-pass system.

The lower hydrolysis rate constant in the recycle system may be the result of product inhibition resulting from the accumulation of hydrolysis products in the recycle system. The concentration of hydrolysis products in the recycle simulation was 266 mg/L while the concentration of hydrolysis products in the single-pass simulation was 37 mg/L.

The lower acidogen growth rate in the recycle system may also be the result of product inhibition cuased by the accumulation of volatile acids in the recycle system. Whether this would also explain the increase in the decay rate is unknown.

In order for hydrolysis products to be product inhibited in the recycle system, it must be presumed that hydrolysis would not be the rate-limiting step under such conditions but that the rate of hydrolysis was controlled to some degree by the rate of removal of hydrolysis products by acidogenesis. Thus, the hypothesis of Eastman and Ferguson (1981) was contradicted in the case of the recycle system.

Although a first comparison of Figures 47 and 49 might seem to indicate that greater difficulty was encountered in fitting the methane production data from the single-pass system as opposed to the recycle system, in fact the converse was true. When the magnitudes of the methane production values were compared, it was evident that the separately fit recycle simulations of methane production showed a difference of as much as 50 L/day from the experimental methane production data, while the single-pass separate fit simulation showed a difference of less tha 2 L/day.

SECTION 8

MODEL MODIFICATIONS

Thus far the results of two alternative approaches to fitting GTLEACH-I to experimental data collected by Schaffer (1986) and Yari (1986) have been considered. In the first case, single-pass and recycle experimental data were fit using a single set of kinetic parameters, while in the second case the two experiments were fit with two separate sets of input parameters. It was concluded that differences among the two landfill environments created by the two leachate management options resulted in the need to fit kinetic constants separately.

It was suggested that the recycle environment resulted in product inhibition of hydrolysis and acidogenic processes, neither of which were included in the GTLEACH-I microbial model. Two approaches to this limitation could be taken. Either development of a separate group of rate constants for recycle systems could be used to account for product inhibition, as was done in the analyses herein, or more complex equations could be developed which explicitly account for product inhibition of hydrolysis and acidogenesis.

Several other difficulties remain to be addresed in applying the GTLEACH-I model to simulation of the Schaffer (1986) and Yari (1986) data.

FLOW MODEL

One of the most pressing issues, not only for application to the Yari (1986) and Schaffer (1986) experiment, but for future modeling efforts as well, is the necessity of a flow model. Because of the regular, weekly rate of moisture application in the Schaffer (1986) and Yari (1986) experiment, a reasonable fit of the data was obtained with GTLEACH-I by averaging the moisture application rate to assume an average daily flow. Such an assumption would not be possible if moisture were applied in a more seasonal manner with accompanying irregular drying and wetting cycles.

Although daily leachate flow measurements were not available to quantify the flow regimes within the single-pass experiment, a tracer study performed by Schaffer (1986) during the acid formation phase and another by Yari (1986) during the methane formation phase, were available. The tracer study performed during the acid formation phase indicated a moisture residence time of nearly five weeks, while the study performed during the methane-formation phase indicated a moisture retention time of 14-17 weeks. This information was not easily converted to leachate flow rate, however, it did indicate that leachate was probably released more quickly after moisture application toward the beginning of the experiment than it was near th end.

In an attempt to capture the range of possible flow regimes encountered during the Schaffer (1986) and Yari (1986) experiments, a series of simulations were conducted which varied the rate of leachate flow among the days of the weekly cycles. The results of causing the entire six liters of water applied to pass though the cell on the day of application, with no flow of leachate on subsequent days until moisture application seven days later, is shown for the single-pass simulation in Figures 50 and 51. This simulation effected no change in the shape of the volatile acid concentration profile, however, it did produce small oscillations in the curve not unlike those observed in the experimental data. The effect on the methane production curve was to slightly increase the rate of methane production--a slight improvement in the fit of the simulation.

Two variable-flow simulations were conducted for the recycle mode: in the first case, all 25 liters of recycled leachate passed through the cell on the day of application; in the second case, flow decreased throughout the week beginning with 10 liters on the day of application, then 5, 4, 3, 2, 1, 0 on the next six days, respectively. Results of these simulations shown in Figures 52 and 53 indicated a more dramatic impact than occurred for the single-pass simulation. Increasing the variability of the flow through the recycle reactor decreased the magnitude of the volatile acid concentration and methane production profiles for the recycle simulation. More interesting was the marked similarity to the data of the large oscillation produced in the volatile acid simulation during the first 50 days.

The preceding simulations provided evidence that a variable flow model could serve to improve the prediction of the GTLEACH-I model. Thus the importance of an accurate flow model should not be underestimated, even for systems where moisture is applied on a regular, cyclical basis. The necessity for an accurate flow model would most certainly increase for systems with irregular rates of moisture application such as would occur in a full-scale landfill.

As indicated previously, most landfills can be characterized as consisting of an unsaturated moisture zone underlain by a saturated zone where leachate collects. For this reason the obvious choice of a flow model for landfill simulation would be an unsaturated flow model which could account for changes in moisture content of the unsaturated zone with rates of moisture infiltration and evapotranspiration. However, there is difficulty in applying unsaturated flow models to landfills due to frequent lack of necessary input parameters such as: unsaturated conductivity with moisture content and time. Further complicating the problem is the fact that all of these parameters change with the age of the landfill as the landfill becomes more consolidated and stabilized.

HYDROLYSIS RATE CONSTANT

A second problem encountered in the previous simulations of the Schaffer (1986) and Yari (1986) data was that the hydrolysis rate constants for all the simulations were several orders of magnitude lower than the value reported by Eastman and Ferguson (1981). Although applicability of such a value, measured in a conventional stirred reactor, to a landfill system can

FIGURE 50

COMPARISON OF CONSTANT FLOW
WITH CYCLICAL VARIABLE FLOW, SINGLE-PASS

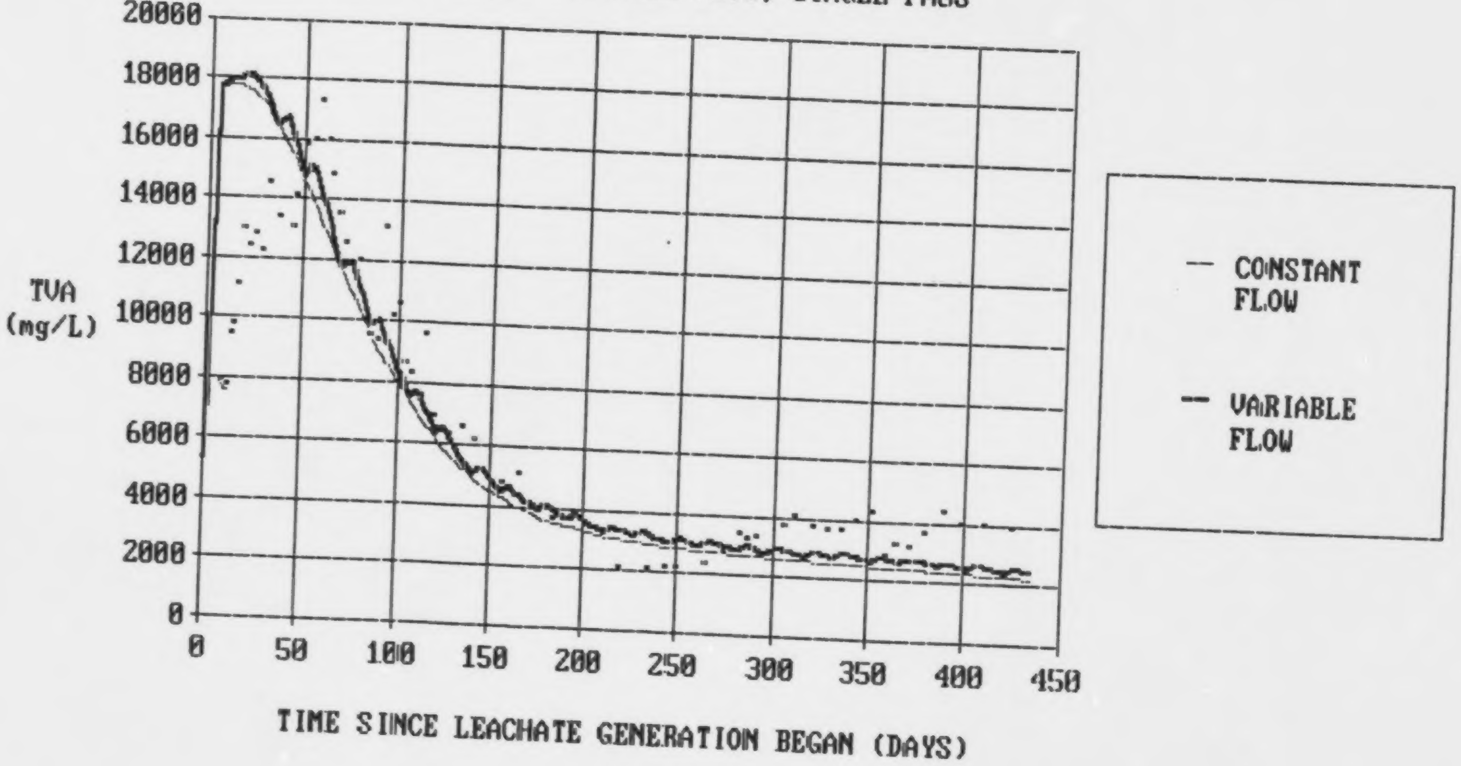


FIGURE 51

COMPARISON OF CONSTANT FLOW
WITH CYCLICAL VARIABLE FLOW, SINGLE-PASS

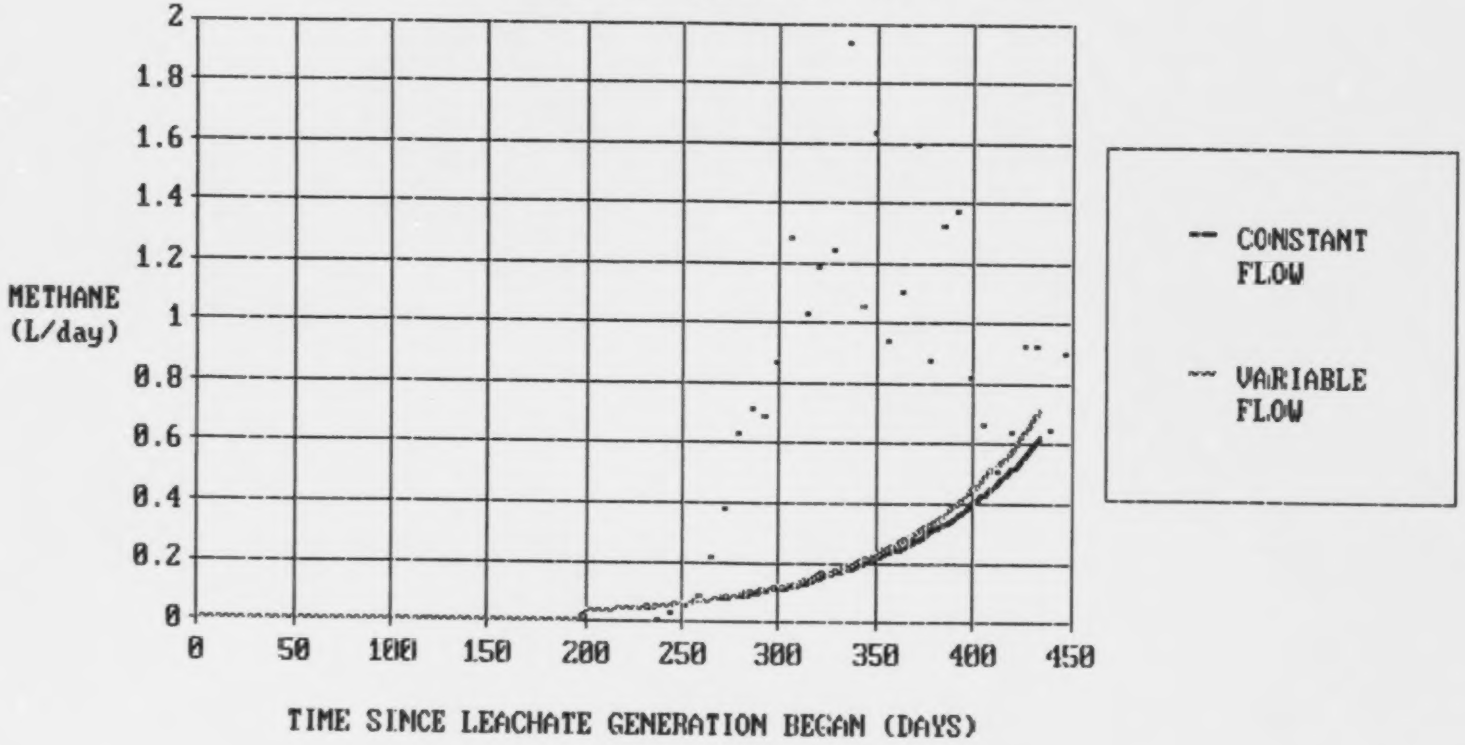


FIGURE 52 ᴜʀE 65

COMPARISON OF EFFECTS OF VARYING DISTRIBUTIONS OF FLOW ON VOLATILE ACID CONCENTRATION, RECYCLE

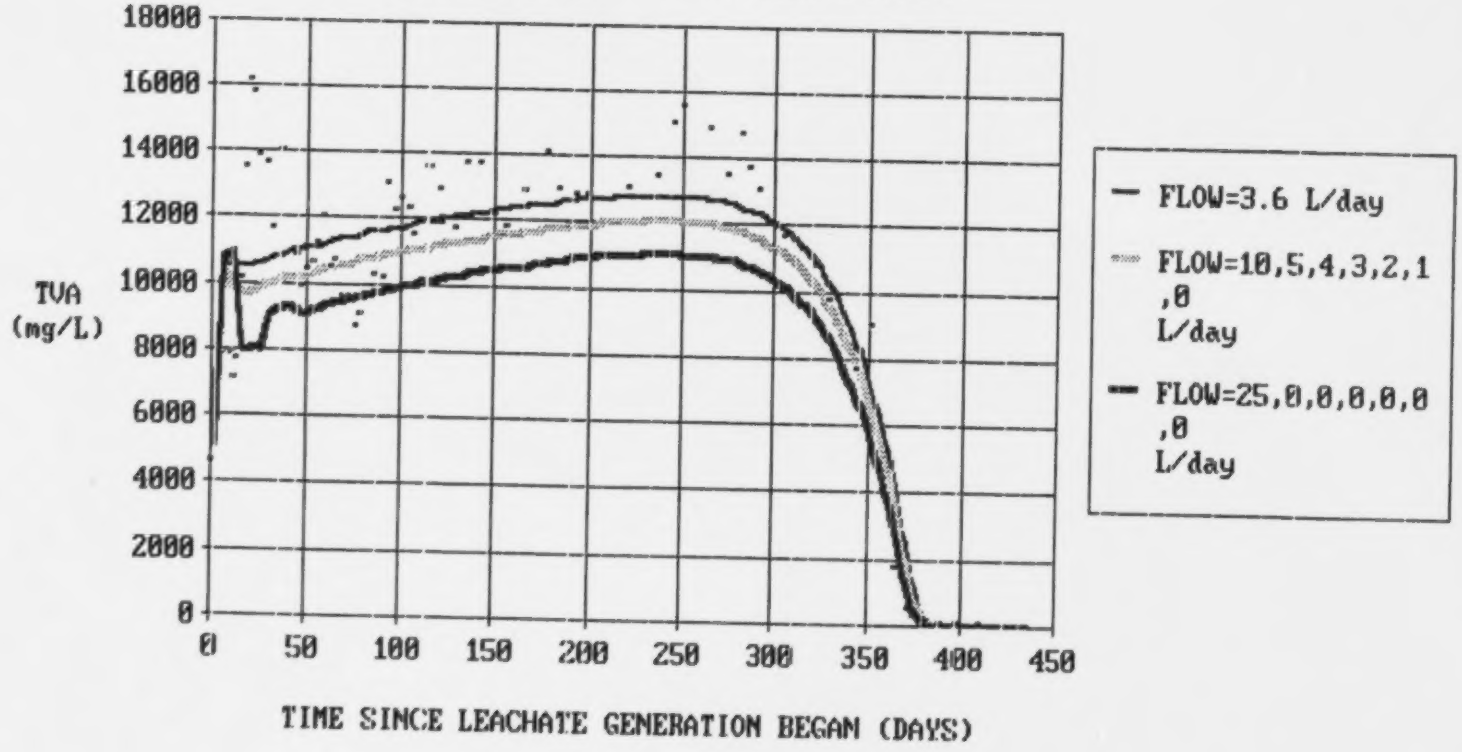
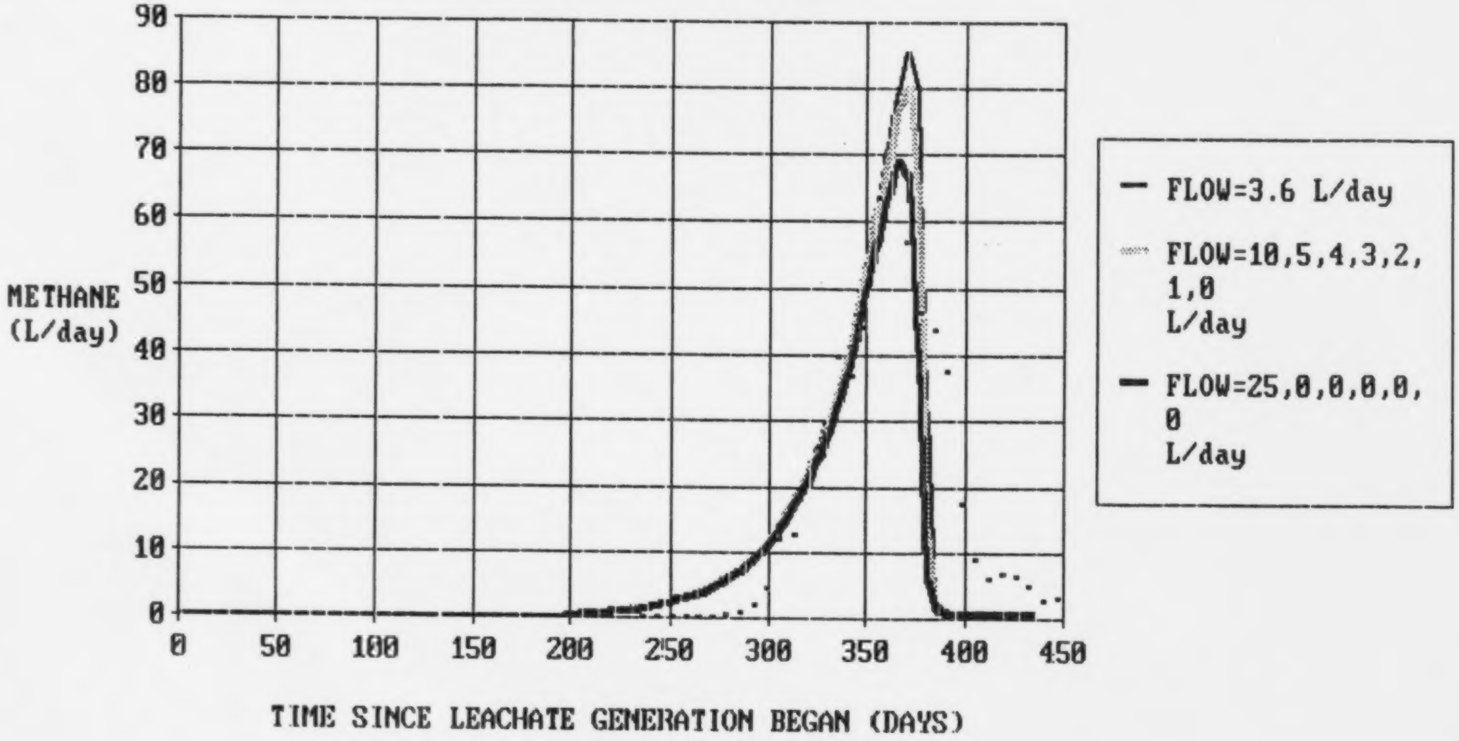


FIGURE 53

COMPARISON OF EFFECTS OF VARYING DISTRIBUTIONS OF FLOW ON METHANE PRODUCTION, RECYCLE



surely be questioned, it is nevertheless valid to question the fitted values as well. Examination of Equation (55) for hydrolysis reveals two components of the hydrolysis reaction rate: the hydrolysis rate constant and the concentration of solid substrate. One possible explanation for the excessively low hydrolysis rate constants fitted to the data may be that there was an overestimation of biodegradable substrate fraction of the solid mass. This fraction is difficult to measure and is frequently omitted from compositional analysis of refuse.

The effects of decreasing the initial degradable mass of solid substrate by order-of-magnitude steps and correspondingly increasing the hydrolysis rate constants are demonstrated in Figure 54, 55, 56 and 57. These changes produced increasingly poorer effects on the fits of both volatile acid and methane production data for the single-pass simulation. In the case of the recycle simulation, decreasing the degradable substrate from 11 kg to 1.1 kg and increasing the hydrolysis rate constant from 0.0005/day to 0.005/day decreased the magnitude of the volatile acid concentration curve, but did not unduly affect the fit of the volatile acid data. This same decrease on the overall production of methane, improving the fit of the methane production data. Thus, no specific conclusion could be drawn as to the validity of the approximation of 20% degradable substrate in the municipal solid waste used in the GTLEACH-I model simulations.

SECONDARY RELEASE OF SUBSTRATE

An upward trend in volatile acid concentration was observed in the single-pass experiment data after approximately 250 days, but not reflected in the simulation. One explanation for such a phenomenon could be that it was the result of a secondary release of substrate. In such a scenario, easily degradable solid substrate began to be exhausted by Day 200 so that volatile acid production decreased. Meanwhile, volatile acids continued to be washed out of the reactor. The resulting depletion of volatile acids in the landfill moisture caused solubilization of the more refractory solid substrate. This process would then result in the production of additional volatile acids and concomitant methane production.

An attempt to simulate this effect was made by increasing the available mass of solid substrate on Day 250. Results of this simulation are shown for the single-pass mode in Figures 58 and 59. A fairly accurate representation of the observed volatile acid and methane gas behavior was produced by this simulation.

DELAY IN ESTABLISHMENT OF ACIDOGENESIS

One final problem encountered in simulation of the Schaffer (1986) and Yari (1986) data was that the peak volatile acid concentration, particularly for the single-pass experiment, appeared later than predicted by the simulation. One possible explanation for this could be that short-circuiting of the moisture flow through the reactor during the first 30 to 75 days of leachate production made less solid substrate available than would be the situation otherwise. To test this hypothesis, the available mass of solid substrate was incrementally increased over the first 50 days of leachate

FIGURE 54

COMPARISON OF EFFECT OF VALUE OF INITIAL XMASSO AND KH ON VOLATILE ACID CONCENTRATION, SINGLE-PASS

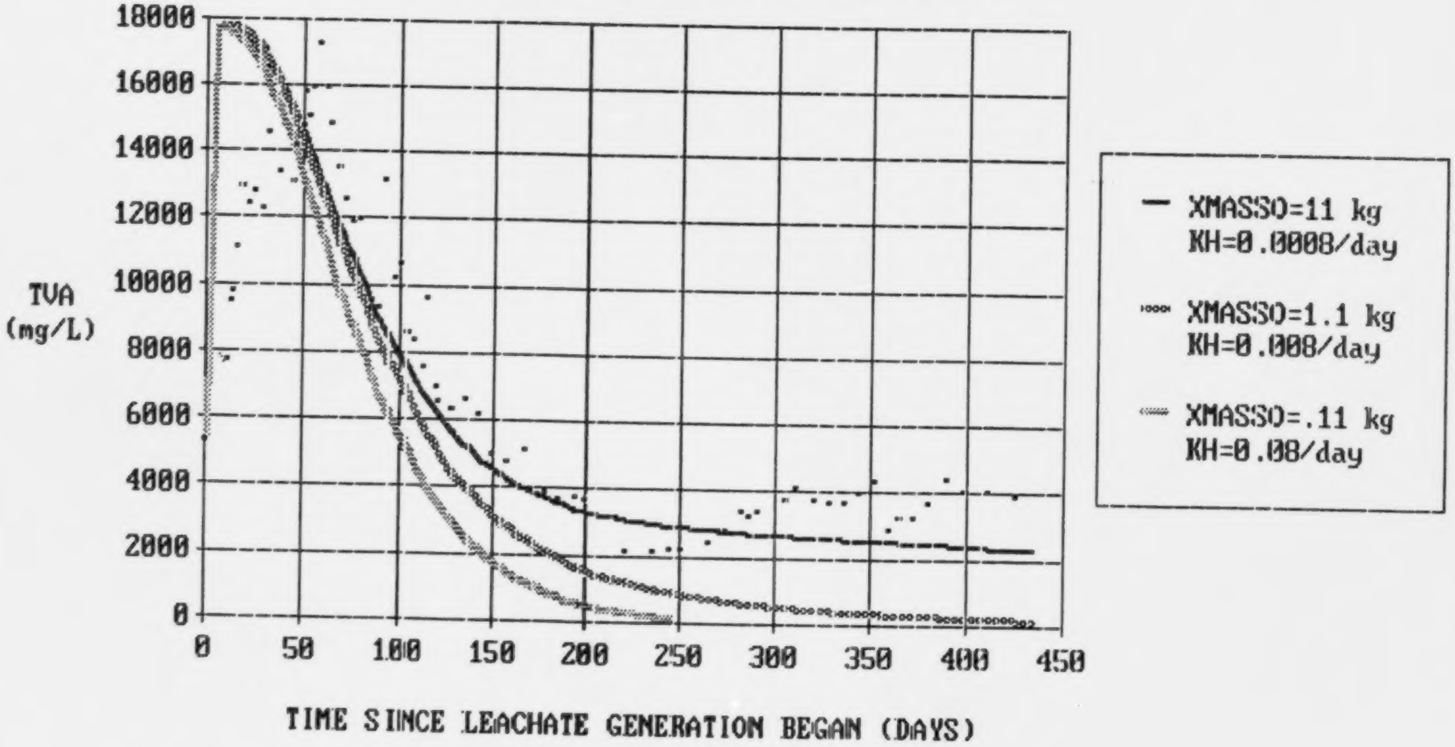


FIGURE 55

COMPARISON OF EFFECT OF VALUE OF INITIAL XMASSO AND KH ON METHANE PRODUCTION, SINGLE-PASS

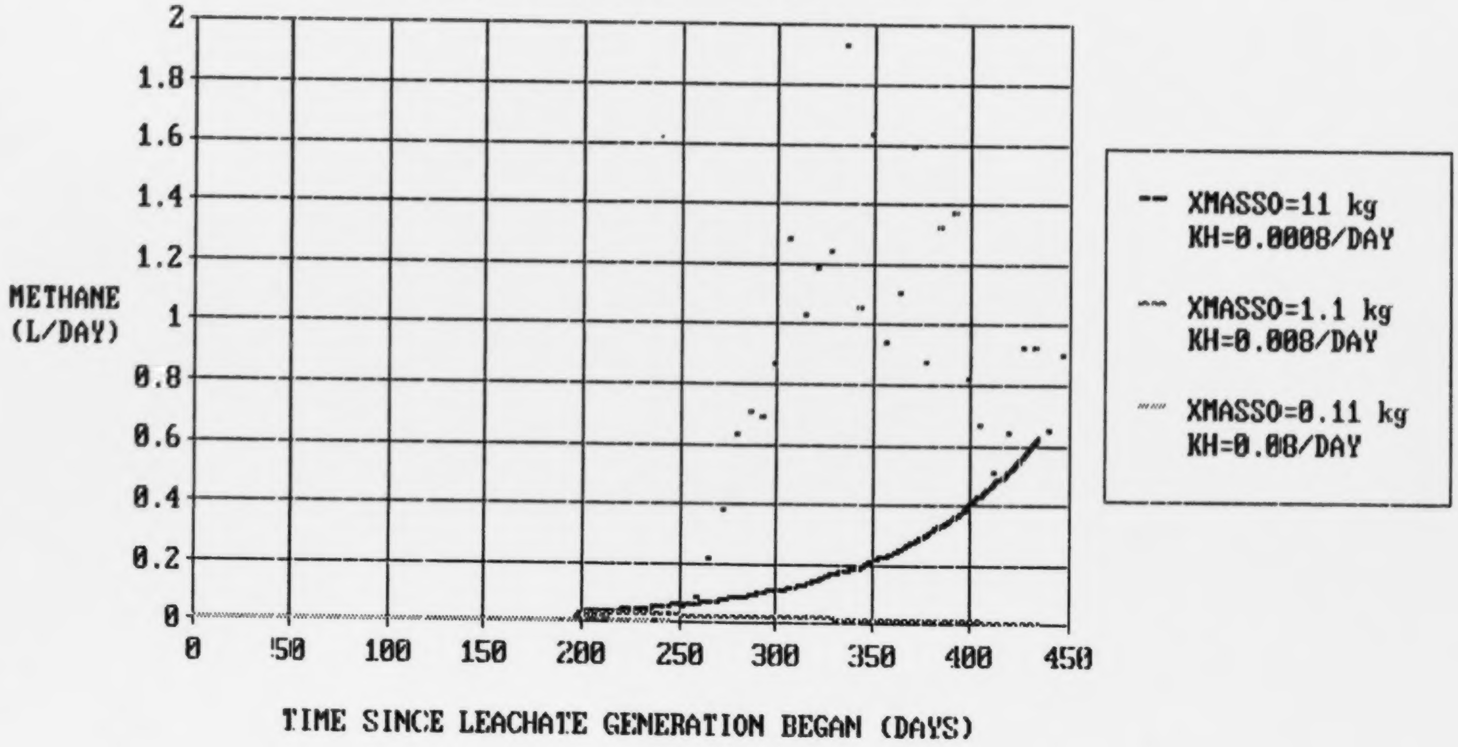


FIGURE 56

COMPARISON OF EFFECT OF VALUE OF INITIAL XMASSO AND KH ON VOLATILE ACID CONCENTRATION, RECYCLE

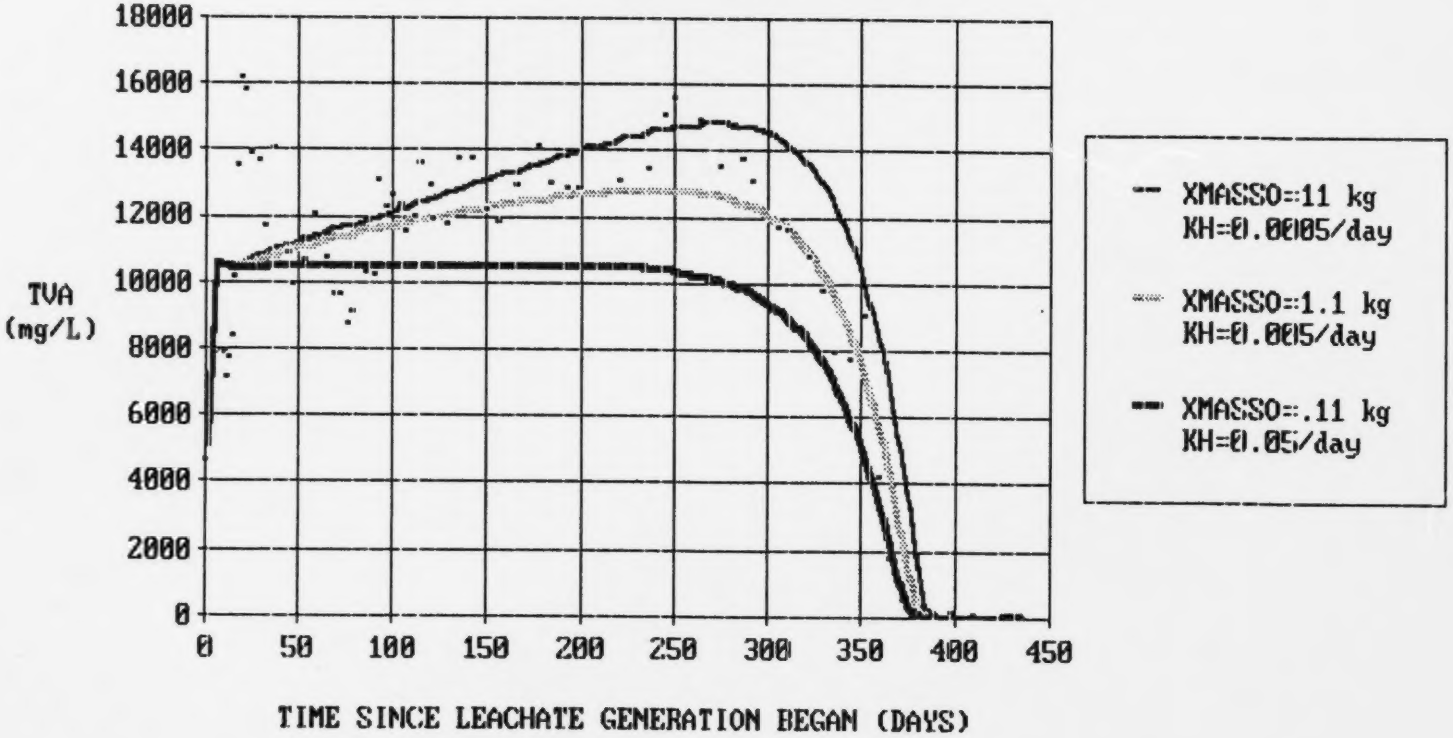


FIGURE 57

COMPARISON OF EFFECT OF VALUE OF INITIAL XMASSO AND KH ON METHANE PRODUCTION, RECYCLE

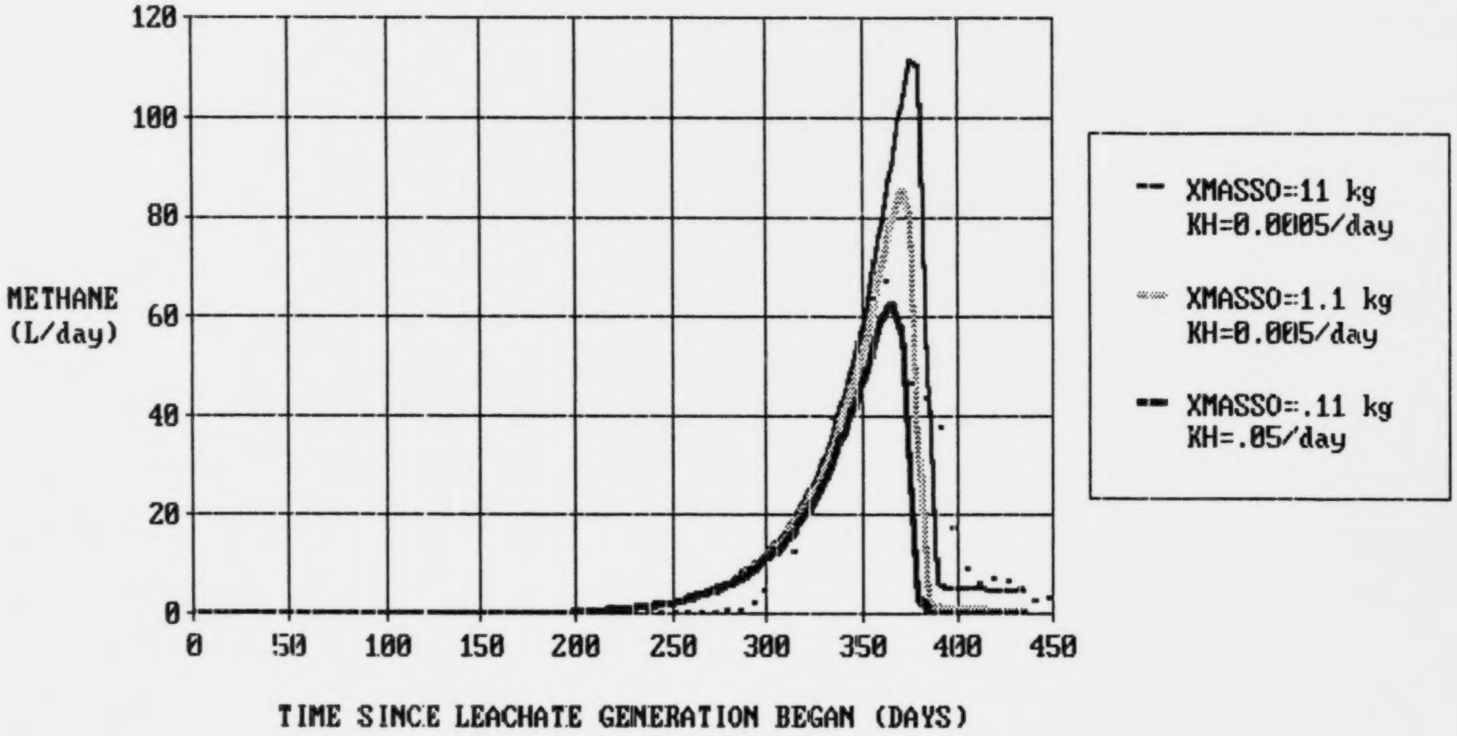


FIGURE 58

EFFECT OF SECONDARY RELEASE OF SUBSTRATE
ON VOLATILE ACID CONCENTRATION, SINGLE-PASS

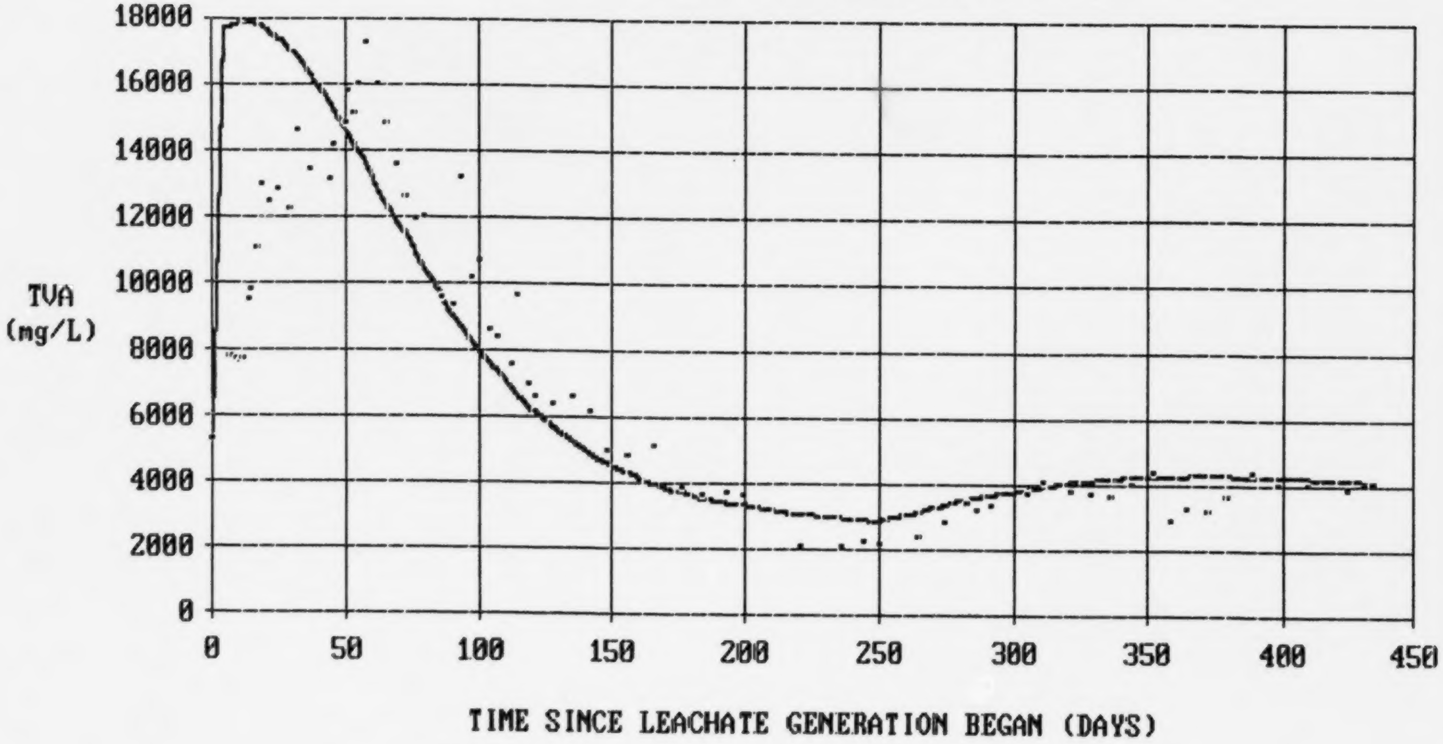
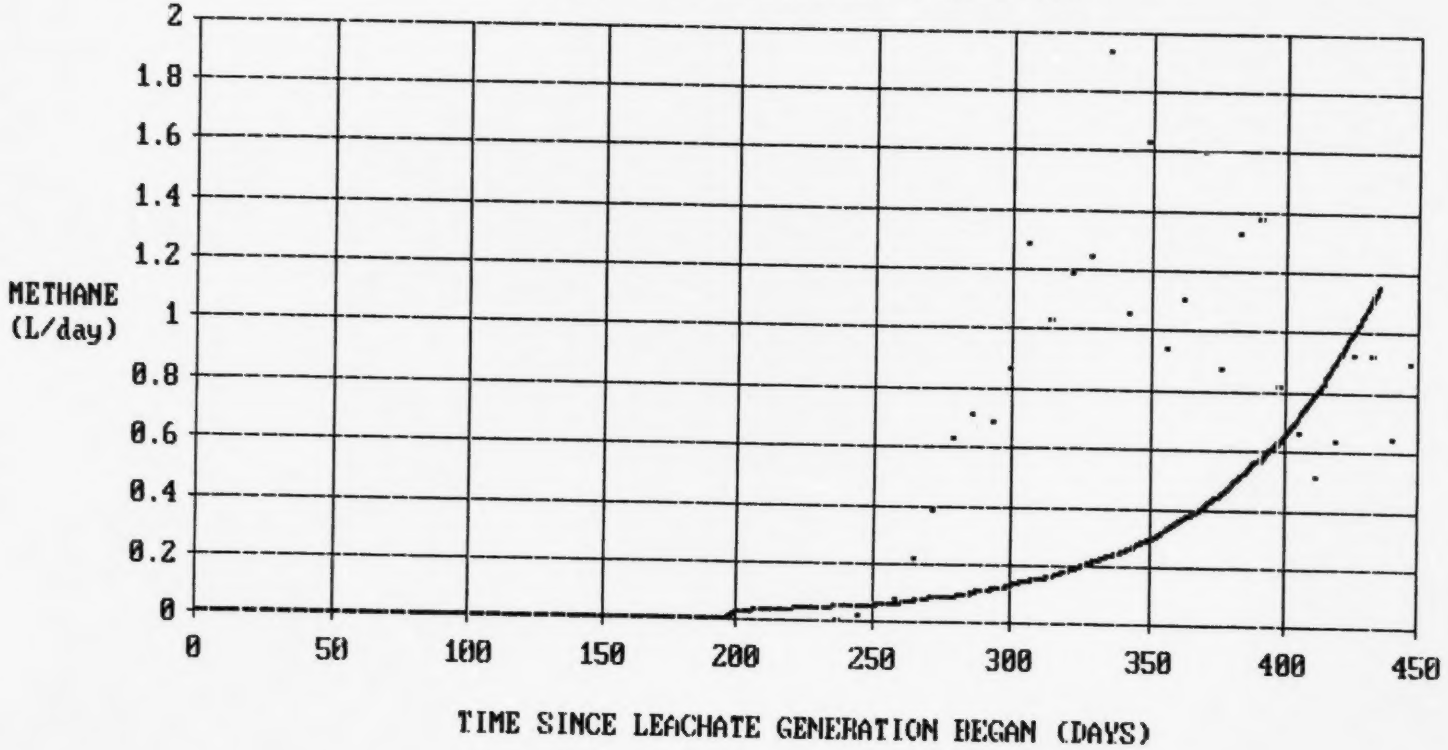


FIGURE 59

EFFECT OF SECONDARY RELEASE OF SUBSTRATE
ON METHANE PRODUCTION, SINGLE-PASS



production to simualte decreasing short-circuiting with time. There was no significant change in the volatile acid concentration curve resulting from this modification for either the single-pass or recycle operational modes. Thus, it seems that the early behavior of the volatile acid concentration profile was not influenced greatly by initial low availability of solid substrate caused by the simulation selected. However, short-circuiting may be involved in a different manner. It is possible that peaks in volatile acid concentrations result when pools of volatile acids isolated within the landfill material are contacted by the flowing moisture during an infiltration event thereby causing a sudden release of volatile acids. Such heterogeneities within the landfill would alter the degree of saturation wtih time and are difficult to predict and model.

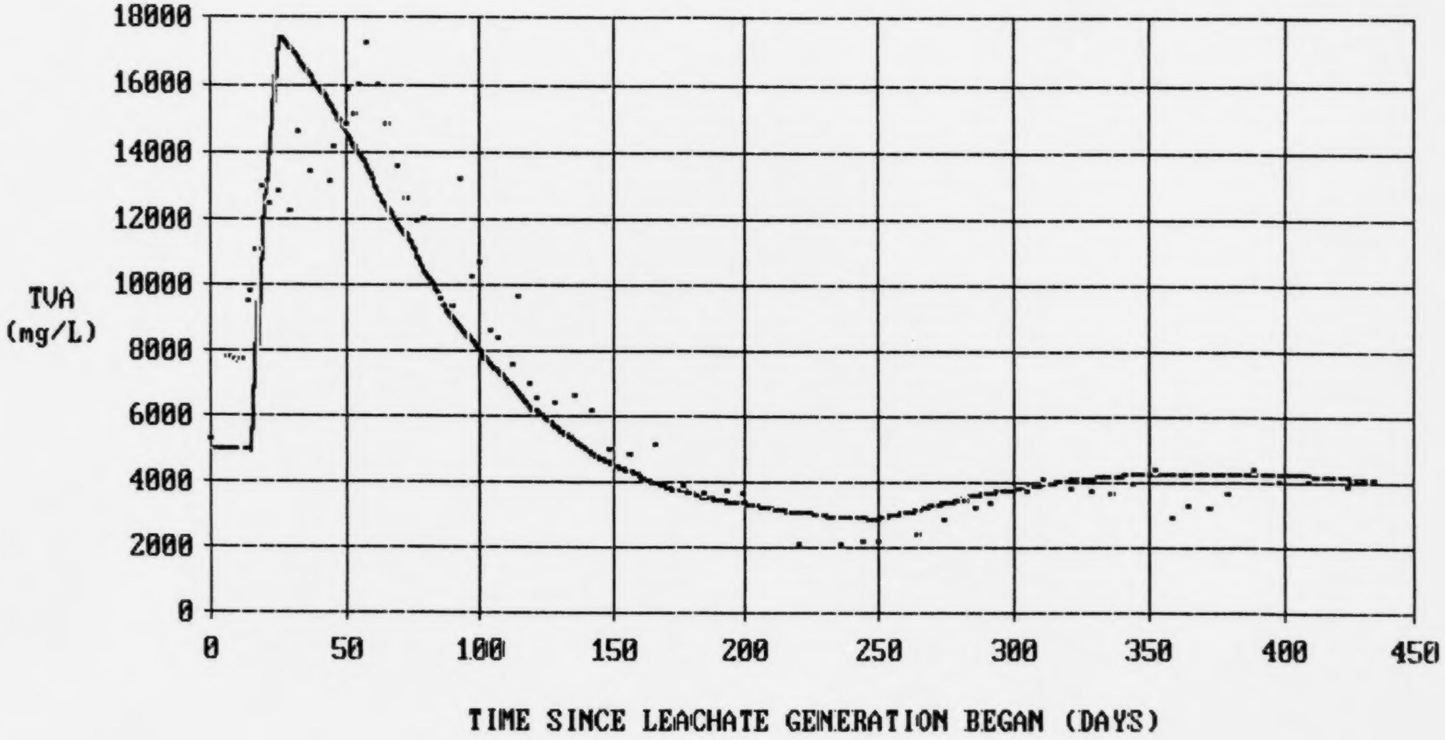
Another explanation for the delayed peaking of the voaltile acid concentration data could be related to the nature of the processes occurring within the population of acidogens. It shoud be expected that even after leachate production begins, there is still much transition occurring within the microbial populations of the landfill cell and that acidogenesis is not established as quickly as modeled in GTLEACH-I. a single Monod equation may be too simplistic to model the changing nature of the relative populations of the various species of acidogenic bacteria as acidogenesis is established.

On solution to this problem is to avoid modeling this transitionary period, but instead to focus modeling efforts on the time of the peak volatile acid concentration and the subsequent volatile acid concentration profile. Shifting the volatile acid concentration peak forward in time was accomplished by suppressing acidogen activity for a specified period of time, e.g., 20 days, as shown in Figure 60. This was also the technique employed in accouting for pH inhibition of methanogens in GTLEACH-I. This technique produced a satisfactory simulation of experimental data but could not be effective without a priori knowledge of the length of the lag inhibition period.

The discussion in this section has presented modifications to GTLEACH-I which would allow its use in a more diagnostic and/or predictive manner. Development of an unsaturated flow model for GTLEACH-I was concluded to be critical to future modeling efforts, particularly when extended to full-scale landfills with intermittent infiltration of moisture. An investigation of the degradable fraction of municipal solid waste, possibly even subcategorization of this fraction (e.g., carbohydrate, protein, lipid and cellulose fractions) would make possible a more accurate model of hydrolysis/solubilization processes. Finally modifications to the microbial equation for acidogenesis and methanogenesis which would accout for product inhibition of acidogenesis and pH inhibition of methanogenesis are suggested.

FIGURE 60

EFFECT OF LAG IN INITIATION OF ACIDOGENESIS
ON VOLATILE ACID CONCENTRATION, SINGLE-PASS



SECTION 9

FITTING GTLEACH-I TO SECOND SET OF EXPERIMENTAL DATA

As a final part of this investigation, the GTLEACH-I model was applied to a second set of experiments conducted several years earlier in the same containers used by Schaffer (1986) and Yari (1986). In experiments by Chang (1982), 66 kg of shredded municipal refuse on a dry basis were packed into 170 liters, in comparison with the 55 kg into 170 liters employed by Schaffer (1986). Initial field capacities established in the Chang (1982) and Schaffer (1986) experiments were similar. The rate of moisture application in the single-pass experiment conducted by Chang (1982) was one liter per week as compared to Schaffer's six liters per week. Chang's recycle experiment included the net addition of 0.5 L per week of deionized water as well as complete recycle of leachate. Therefore, the average weekly flow through the recycle cell increased by 0.5 L/wk throughout the experiment. Another major difference between the Chang (1982) experiment and the Schaffer (1986) and Yari (1986) experiments was that although methanogenesis was inhibited during the first 20 to 50 days after leachate generation began, it was established without seeding in the Chang (1982) experiment.

The single-pass and recycle volatile acid profiles for the two Chang (1982) experiments are shown in Figure 61. A comparison of this figure with Figure 62 showed that there was much greater similarity between the volatile acid concentration profiles for the two modes in the Chang experiment than the Schaffer (1986) and Yari (1986) experiments. This was due to diminished effects of washout resulting from lower moisture application rates in the Chang (1982) single-pass experiment which caused the single-pass system to behave more like the recycle system than was the case in the Schaffer (1986) and Yari (1986) experiments.

The results of simulation of the volatile acid concentration profile for the Chang (1982) single-pass experiment employing the same kinetic parameters as applied in the separate simulation of Schaffer (1986) and Yari (1986) single-pass data are presented in Figures 63 and 64. There was an overestimation in the simulation of approximately 200 days for the time to virtually complete removal of the volatile acids from the system. Likewise, there was a gap of nearly 100 days between the peak in methane production shown in the experimental data and that of the simulation. The magnitude of the methane production curve was also grossly overpredicted. (Problems with overprediction of methane production may have been due in part to experimental difficulties, possibly a leak in the landfill cell.)

Adjustments to the input kinetic parameters were made in order to improve the fit of GTLEACH-I simulations to the Chang (1982) single-pass

FIGURE 61

COMPARISON OF SINGLE-PASS AND RECYCLE VOLATILE ACID CONCENTRATION
DATA COLLECTED BY CHANG (1982)

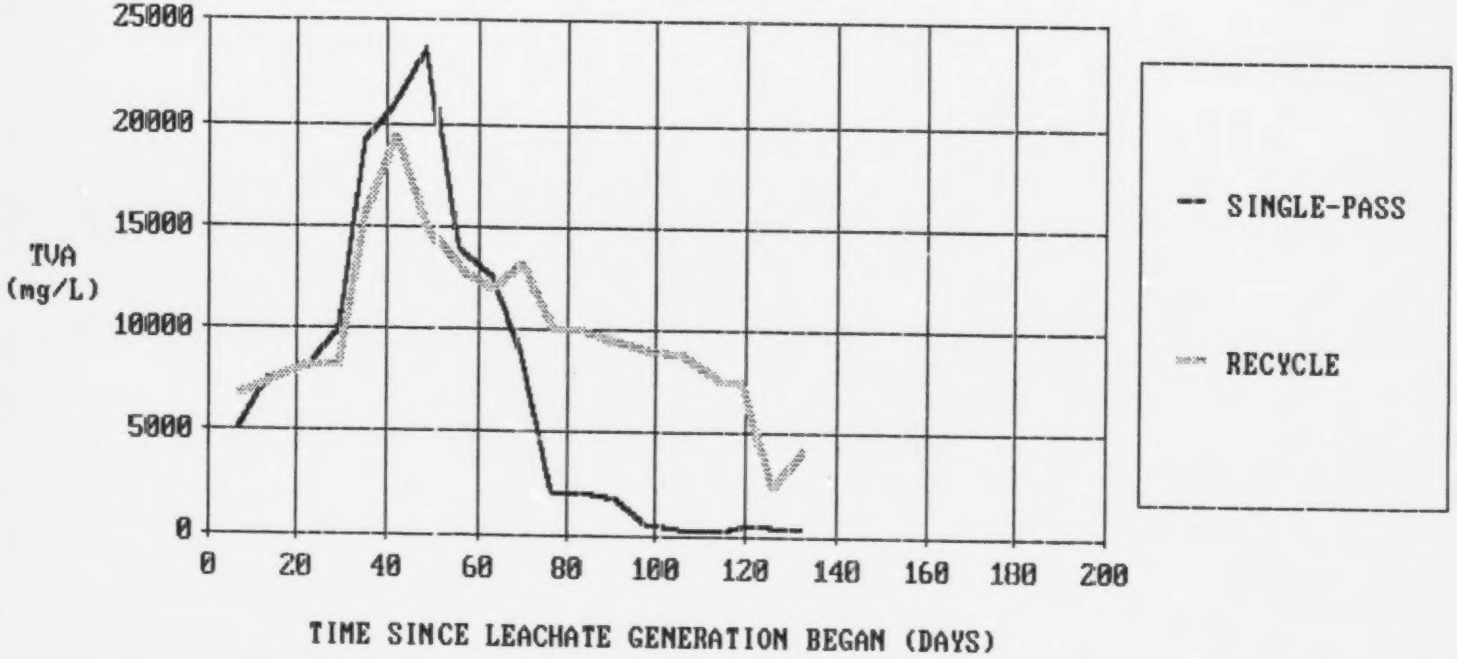


FIGURE 62

COMPARISON OF SINGLE-PASS AND RECYCLE VOLATILE ACID CONCENTRATION DATA COLLECTED BY SCHAFFER (1986) AND YARI (1986)

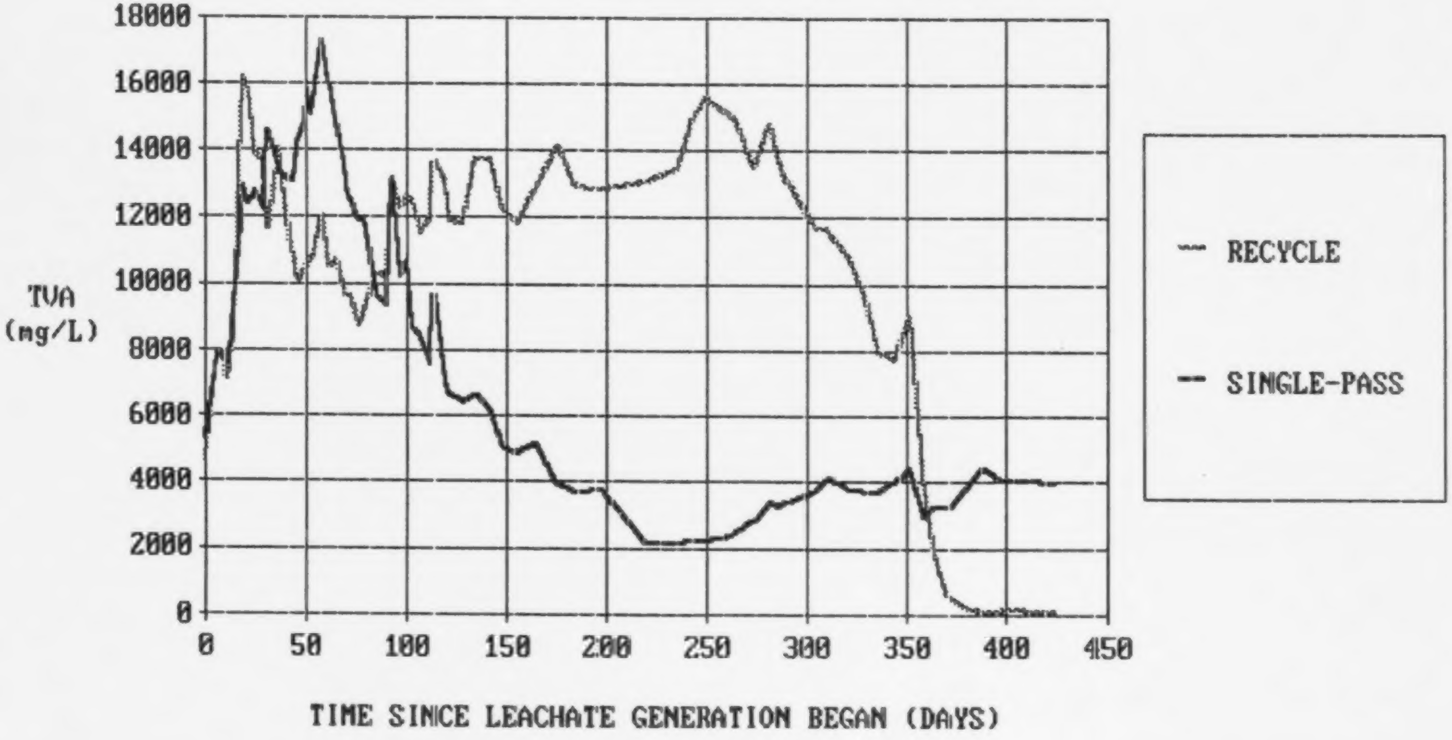


FIGURE 63

COMPARISON OF SIMULATED VOLATILE ACID CONCENTRATION
(USING PARAMETERS FROM SCHAFFER (1986) AND YARI (1986))
WITH EXPERIMENTAL DATA BY CHANG (1982), SINGLE-PASS

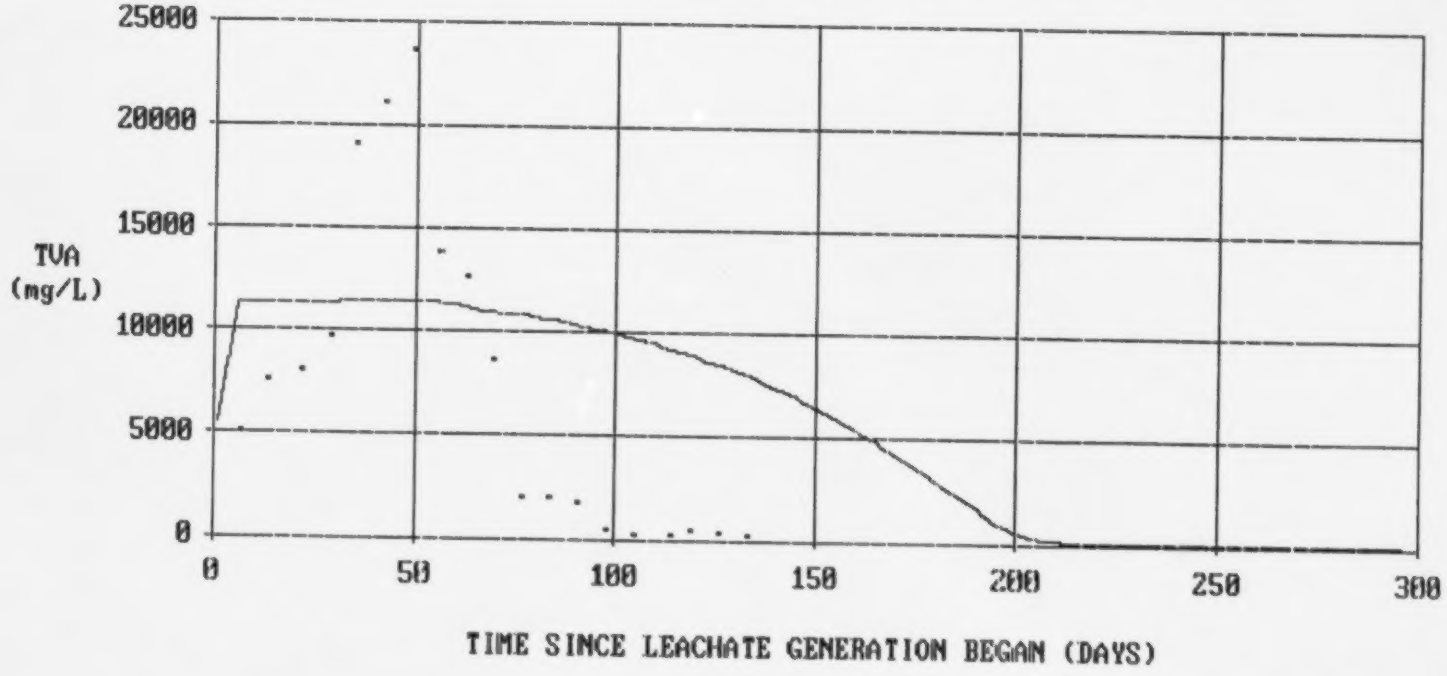
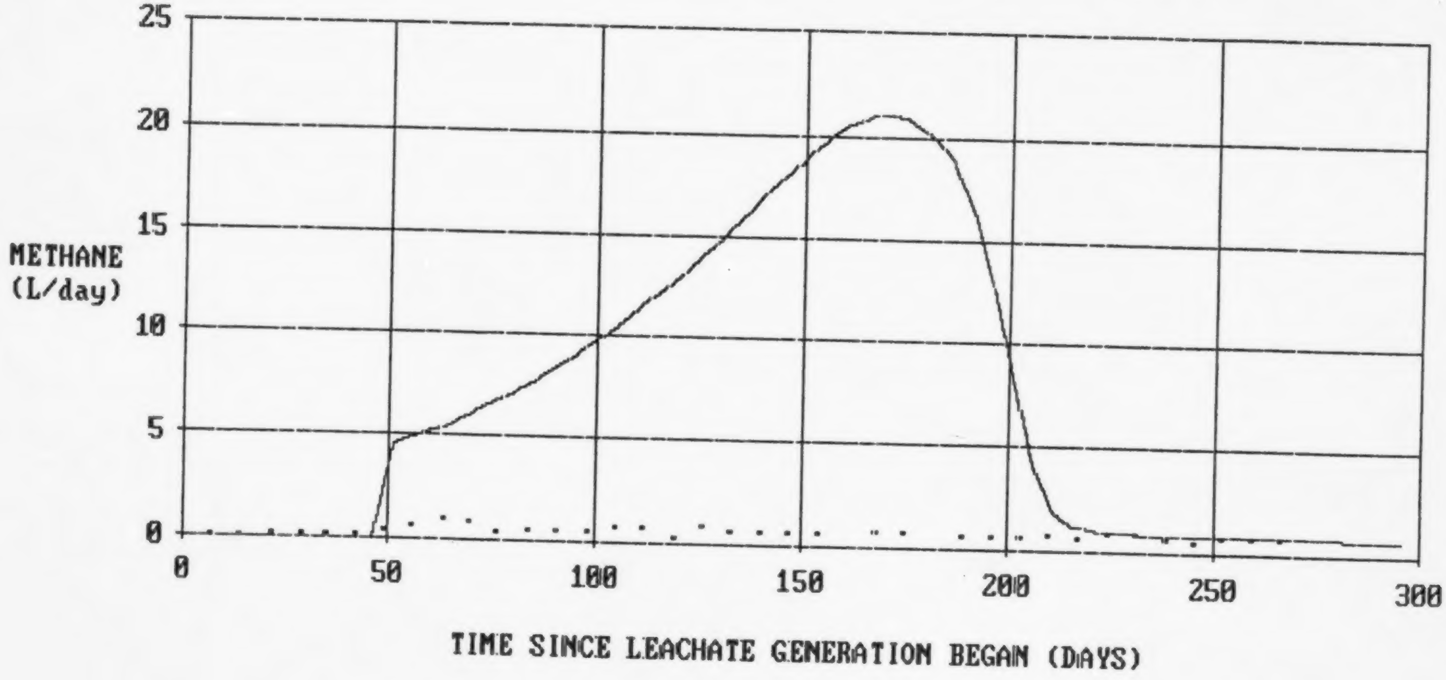


FIGURE 64

COMPARISON OF SIMULATED METHANE PRODUCTION
(USING PARAMETERS FROM SCHAFFER (1986) AND YARI (1986))
WITH EXPERIMENTAL DATA BY CHANG (1982), SINGLE-PASS



data. The results of sequential adjustments in the parameters are demonstrated in Figures 65 through 67. In Figure 65, the maximum substrate utilization coefficient for methanogens, Q_M, was increased from 1.9 to 2.5/day resulting in a shortening of the time to removal of volatile acids. In Figure 66, the lag time before initiation of methanogenesis was decreased from 50 to 10 days, further shortening the time to removal of volatile acids. Finally, in Figure 67 the yield coefficient for methanogens was increased from 0.02 to 0.04, sharply increasing the rate at which volatile acids were removed.

The results of simulation of the Chang (1982) recycle experiment employing the kinetic parameters from the separate fit of the Schaffer (1986) and Yari (1986) data are presented in Figures 68 and 69. These results show a somewhat improved fit than was the case for the single-pass simulation. There was a 50-day overestimation in the time to removal of volatile acids. The peak in methane production was also 50 days later than in the experimental data. In addition, the peak in methane production still continued to be grossly overestimated.

Adjustments of parameters were made to improve the fit of GTLEACH-I simulation of the Chang (1982) recycle data. The effects of increasing the maximum substrate utilization rate for methanogens, Q_M, from 1.9/day to 2.2/day are shown in Figures 70 and 71. The result is to shorten the time to removal of volatile acids and correspondingly shorten the time to peak methane production. A subsequent change in initial hydrolysis product concentration, SHO, from 20,000 mg/L to 15,000 mg/L produced the results shown in Figures 72 and 73. The effects were to slightly decrease the magnitudes of both volatile acid concentration and methane production profiles. The final result was a reasonably good fit of the volatile acid concentration data, however, the magnitude of the methane production curve continued to be overestimated, possibly explained by leaks in the experimental cells which may have resulted in failure to capture the total gas volume generated.

The difference between the single-pass kinetic parameters fit to the Schaffer (1886) and Yari (1986) data and those fit to the Chang (1982) data were:

- the maximum substrate utilization rate for methanogens was 1.9/day for the Schaffer (1986) and Yari (1986) simulation, while for the Chang (1982) simulation it was 2.5/day; and
- the yield coefficient for methanogens in the Schaffer (1986) and Yari (1986) simulation was 0.02 and for the Chang (1982) simulation it was 0.04.

This resulted in an overall increase by 2.5 times in the maximum growth rate constant for methanogens of the Chang (1982) simulation over the Schaffer 91986) and Yari (1986) simulation. This may reflect differences in the inhibitory natures of the two environments. Due to higher rates of moisture application in the Schaffer (1986) and Yari (1986) experiments, the landfill to be in contact with the low pH leachate longer with more likelihood of stronger inhibition.

FIGURE 65

COMPARISON OF SIMULATED VOLATILE ACID CONCENTRATION WITH EXPERIMENTAL DATA BY CHANG (1982), SINGLE-PASS QM=2.5/DAY

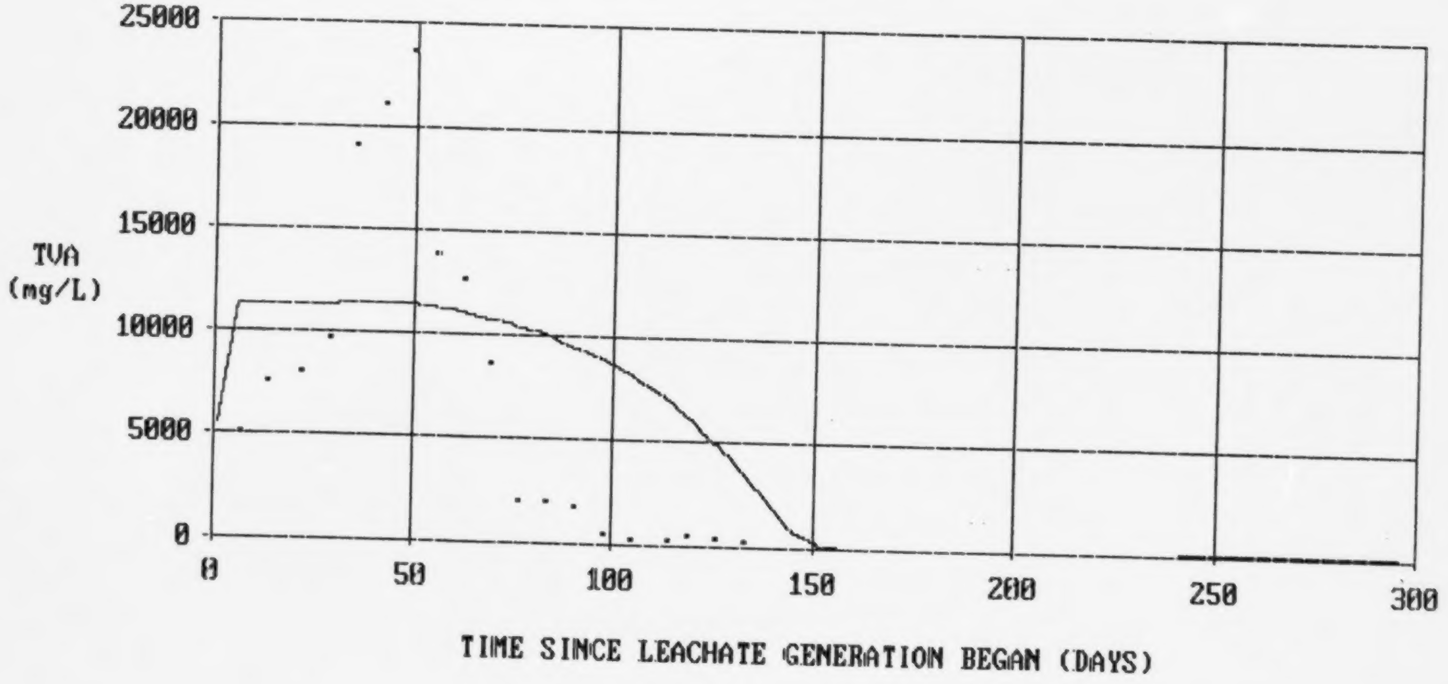


FIGURE 66

COMPARISON OF SIMULATED VOLATILE ACID CONCENTRATION WITH EXPERIMENTAL DATA BY CHANG (1982), SINGLE-PASS
QM=2.5/DAY, LAG=10 DAYS

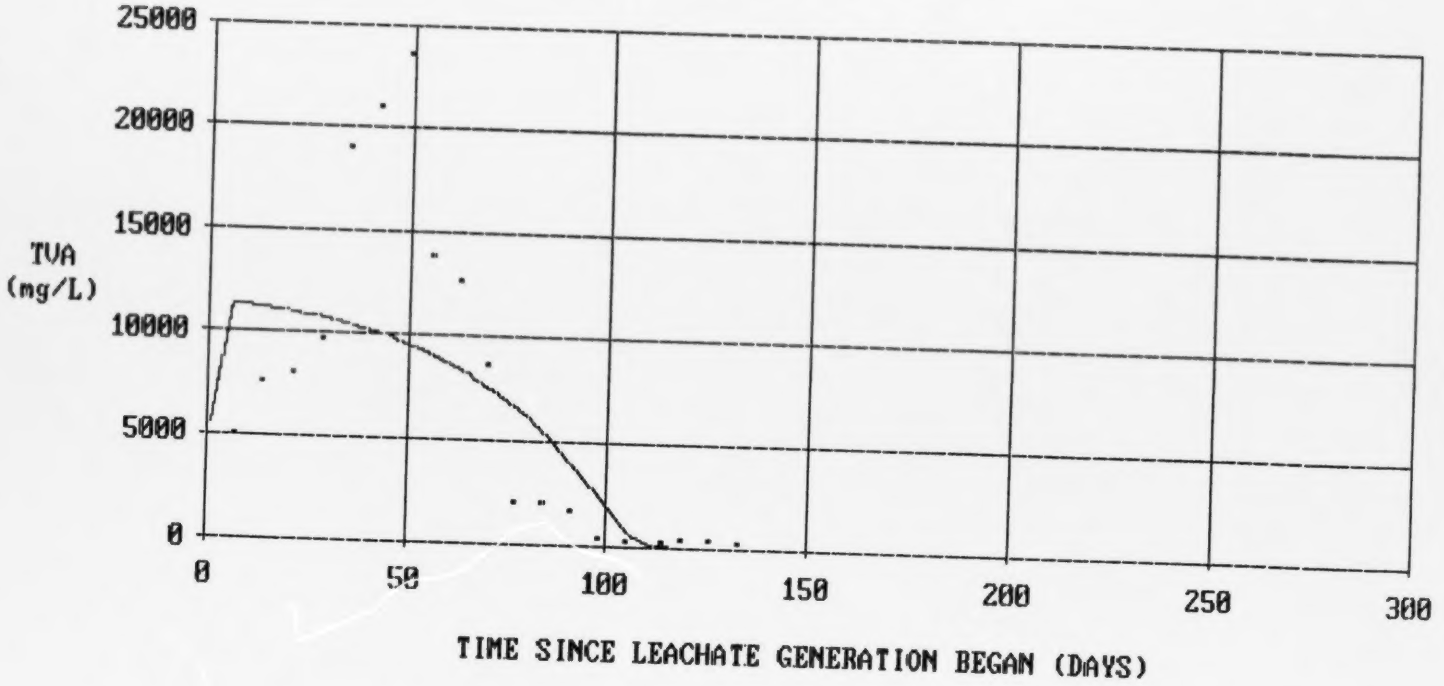


FIGURE 67

COMPARISON OF SIMULATED VOLATILE ACID CONCENTRATION WITH EXPERIMENTAL DATA BY CHANG (1982), SINGLE-PASS
QM=2.5/DAY, Y=0.04, LAG=10 DAYS

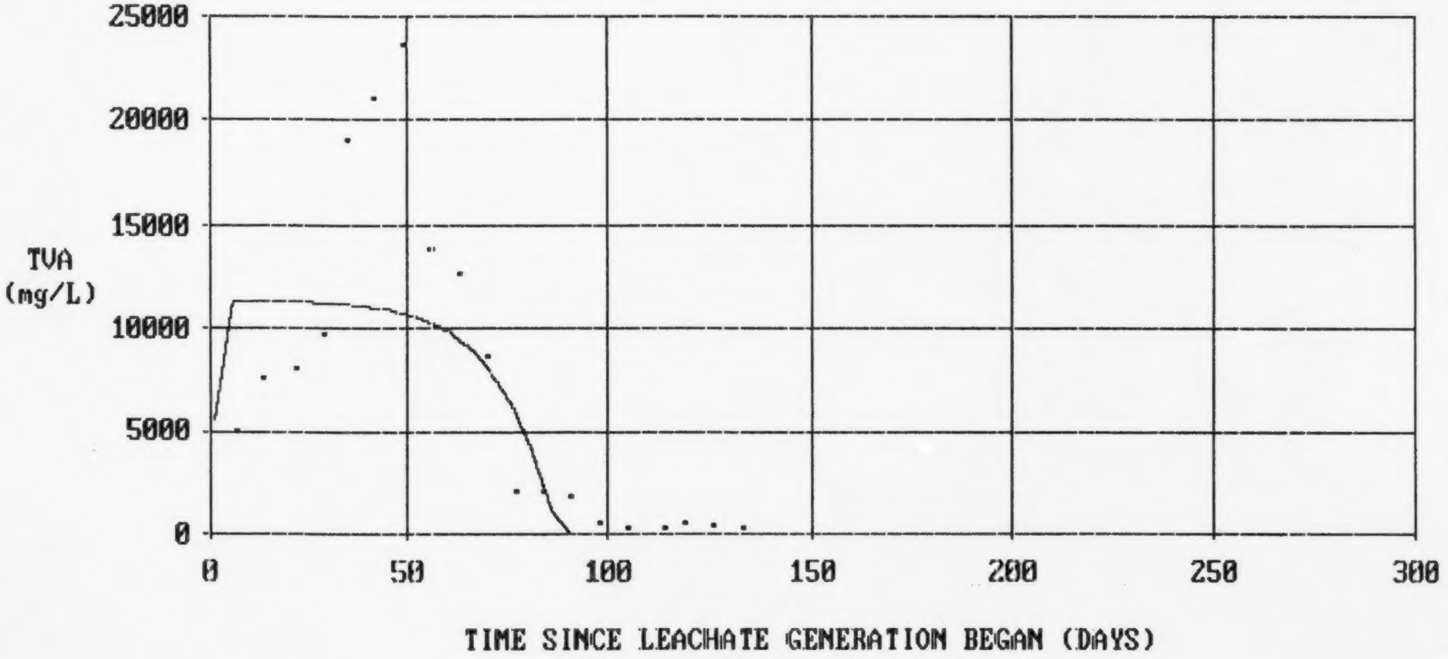


FIGURE 68

COMPARISON OF VOLATILE ACID CONCENTRATION SIMULATION (USING PARAMETERS FROM SCHAFFER (1986) AND YARI (1986)) WITH EXPERIMENTAL DATA BY CHANG (1982), RECYCLE

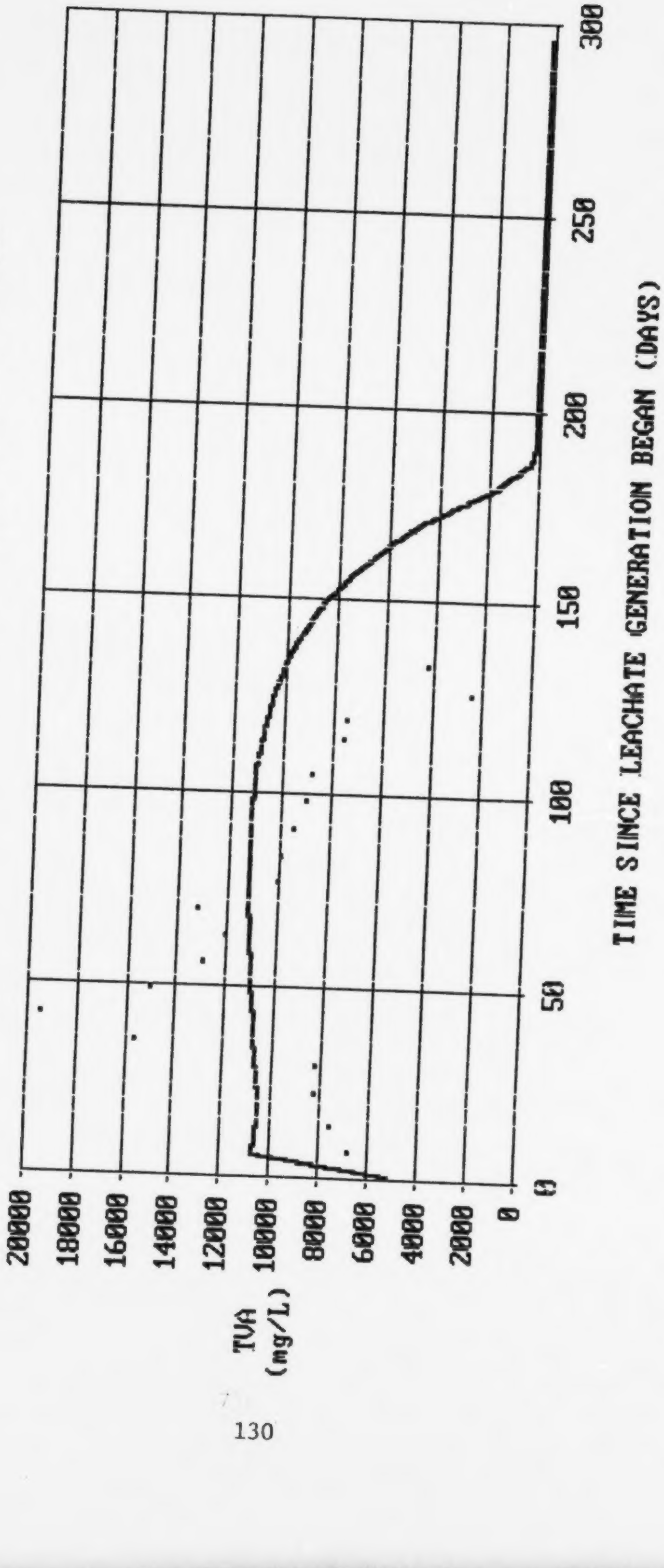


FIGURE 69

COMPARISON OF SIMULATED METHANE PRODUCTION
(USING PARAMETERS FROM SCHAFFER (1986) AND YARI (1986))
WITH EXPERIMENTAL DATA BY CHANG (1982), RECYCLE

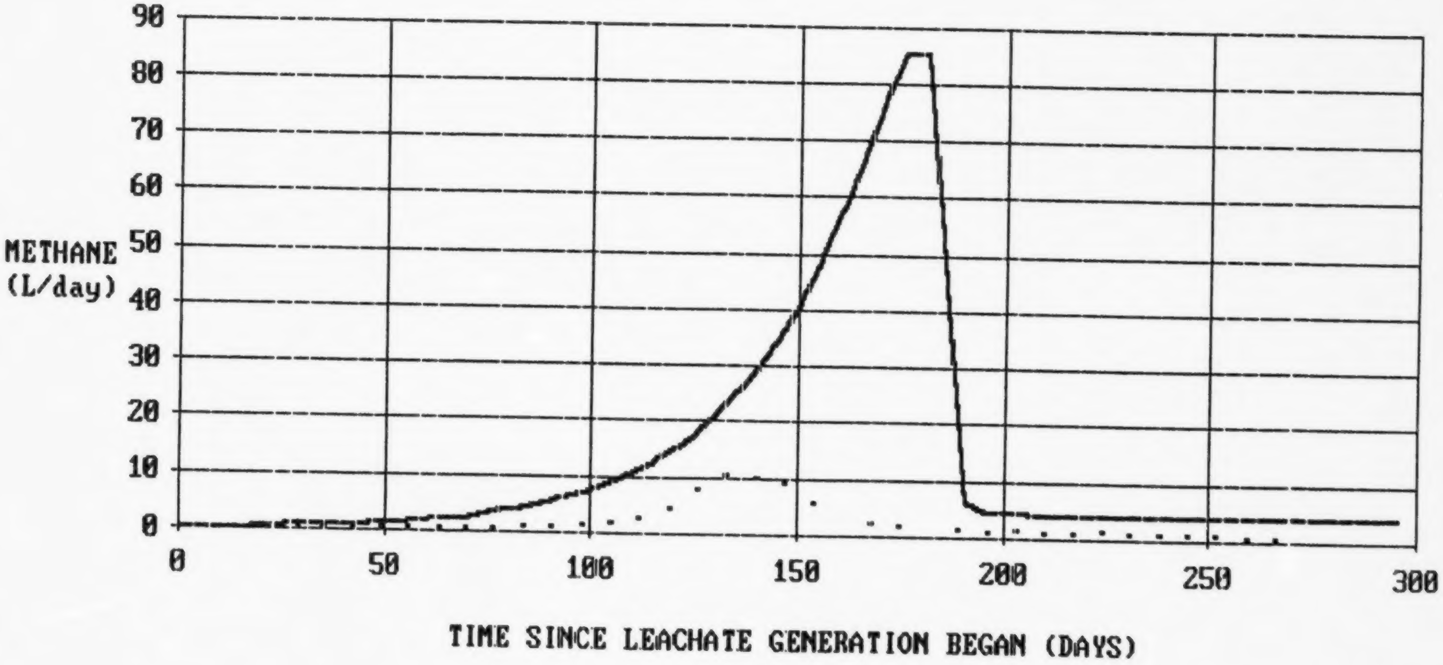


FIGURE 70

COMPARISON OF SIMULATED VOLATILE ACID CONCENTRATION
WITH EXPERIMENTAL DATA BY CHANG (1982), RECYCLE
QM=2.2/day

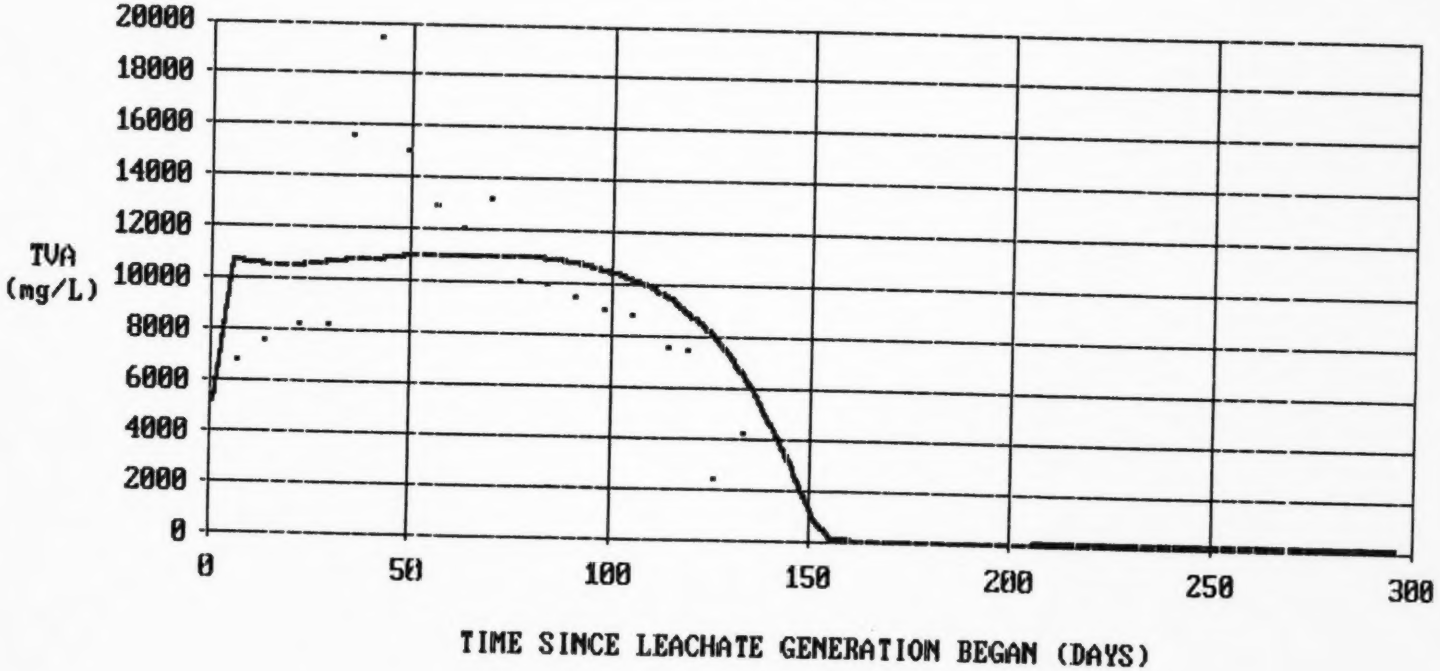


FIGURE 71

COMPARISON OF SIMULATED METHANE PRODUCTION
WITH EXPERIMENTAL DATA BY CHANG (1982), RECYCLE
QM=2.2/day

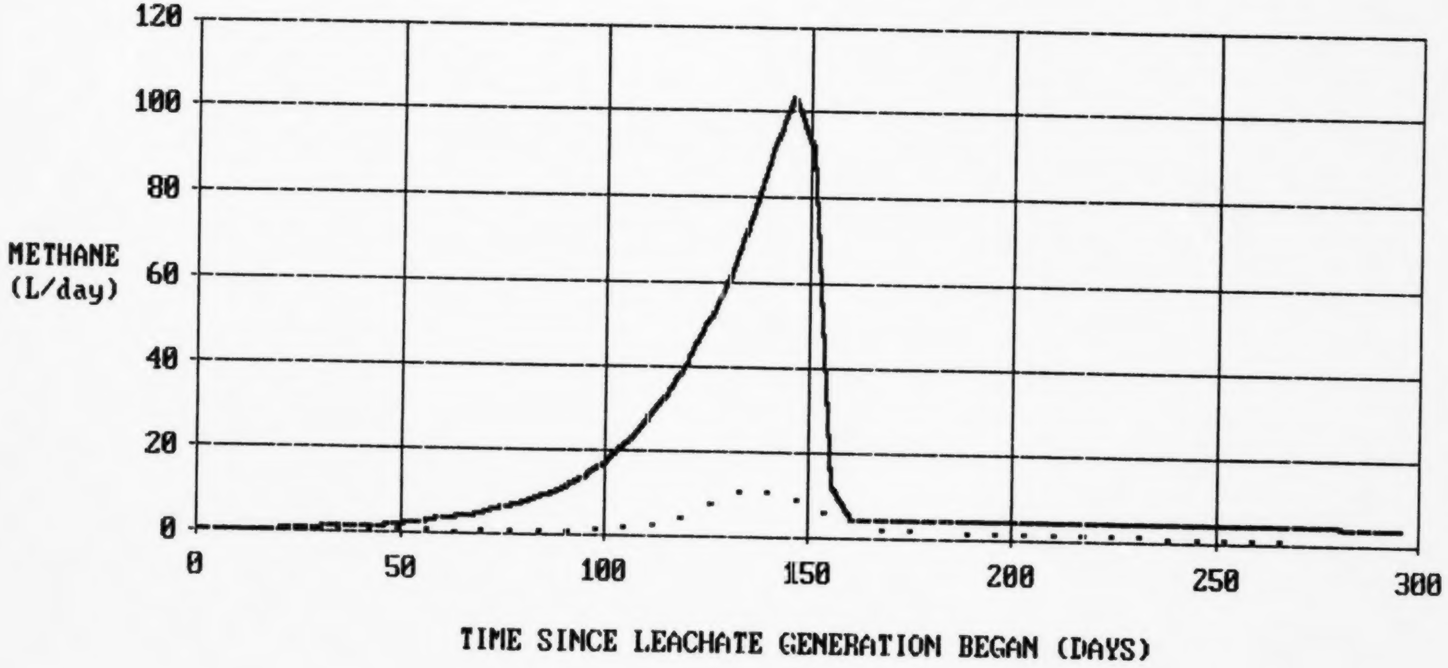


FIGURE 72

COMPARISON OF SIMULATED VOLATILE ACID CONCENTRATION WITH EXPERIMENTAL DATA BY CHANG (1982), RECYCLE
QM=2.2/day, SH0=15,000 mg/L

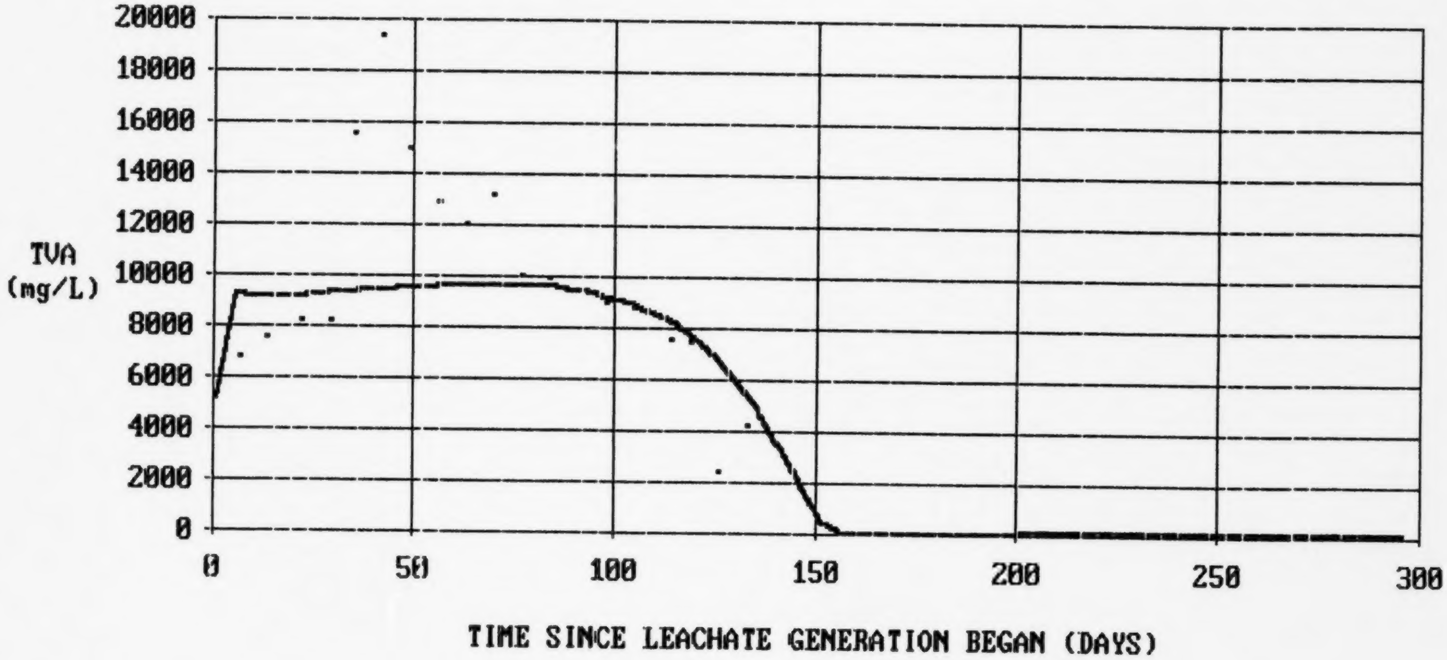
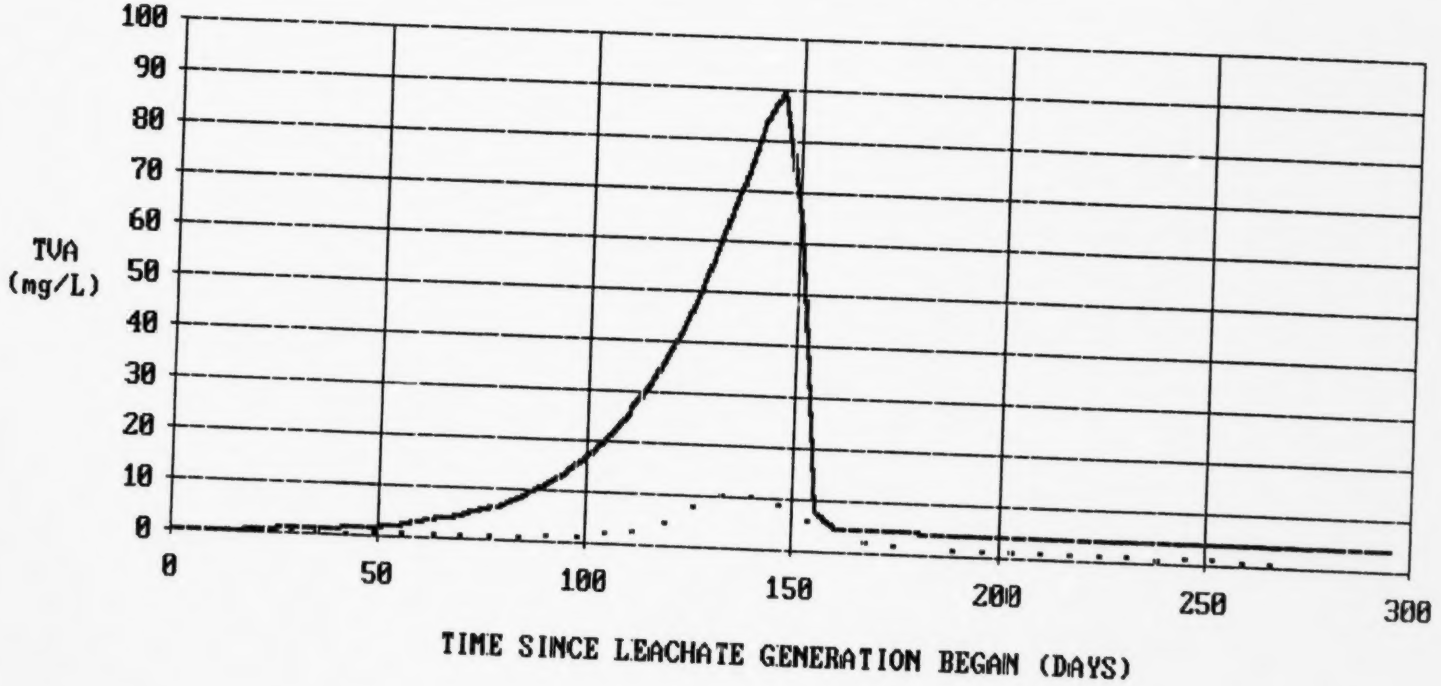


FIGURE 73

COMPARAISON OF SIMULATED METHANE PRODUCTION
WITH EXPERIMENTAL DATA BY CHANG (1982), RECYCLE
QM=2.2/day, SHO=15,000 mg/L



The only difference between the kinetic parameters fit to the Schaffer (1986) and Yari (1986) recycle experiments and those fit to the Chang (1982) recycle experiment was that the maximum substrate utilization rate for methanogens was 1.9/day for the Schaffer (1986) and Yari (1986) simulations and 2.2/day for the Chang (1982) simulation. This again may be related to the increase in moisture application rates of the Schaffer (1986) and Yari (1986) experiments which increased the opportunity for contact of the methanogens with the low pH leachate thereby increasing inhibition. If indeed this was the explanation for the difference in simulated maximum growth rate for methanogens in the two experiments, it must be inferred that simply lagging the initiation of methanogenesis was not sufficient for modeling pH inhibition. The lag was an oversimplification of the pH inhibition process. A more accurate representation of the process would require a relationship between moisture movement per unit volume of the landfill and the degree of inhibition of the methanogenic population, i.e., an unsaturated flow model coupled with a biological model containing an inhibition function.

Thus the only major difference between the microbial kinetics observed in the Schaffer (1986) and Yari (1986) experiments and the Chang (1982) experiments, in both recycle and single-pass cases, was in the methanogen growth rate constant. The hydrolysis/solubilization and acidogenesis constants were consistent between the two experiments. This lends strong support to the three-step model and provides impetus for improvement of the inhibition function at least in the third step.

SECTION 10

SUMMARY AND DISCUSSION OF MODEL GTLEACH-I

The model presented in this report is a three-step simulation of the general anaerobic microbial processes occurring within landfill systems. A reasonable model of volatile acid production and removal has been presented, laying the foundation for future work expanding into the inorganic aspects of landfill stabilization and eventually into a model of pH and pE which would greatly enhance prediction of the fate of hazardous organic constituents co-disposed with municipal waste in landfills. A major accomplishment of the model was the application of kinetic constants for hydrolysis/solubilization and acidogenesis in the simulation of two sets of experimental data having substantially different moisture application rates. The model was also shown to be a useful diagnostic tool in examining differences in microbial populations resulting from different leachate management options.

Two major areas of improvement are recommended for GTLEACH-I. The first improvement would be in the microbial model of methanogeneis, since the simplistic lag model put forth in this work was insufficient to account for pH inhibition of methanogenesis. Development of a pH inhibition model would help to correlate degree of inhibition with pH and the rate of moisture flow through the landfill.

Beyond development of an improved model of pH inhibition of methanogens, the other major area of improvement to GTLEACH-I lies in modeling of moisture flow within landfills. Comparison of constant flow versus intermittent flow dramatized the importance of accurate flow simulation in modeling landfill stabilization. Eventual coupling of the microbial model developed in GTLEACH-I with a companion unsaturated flow model would do much to enhance the ability of the model to simulate behavior of landfills during stabilization.

The inability of the model to simulate fluctuations in volatile acid concentrations during the first 50 or so days of landfill life following leachate generation was attributed to effects of short-circuiting and the transitional nature of the microbial populations during this period. Due to the difficult, yet short-lived nature of this problem, it is recommended that immediate modeling efforts focus on the subsequent volatile acid profile.

Further research into the composition of municipal solid waste in terms of degradable fraction, as well as carbohydrate, protein, lipid and cellulose subfractions is also recommended. Acquisition of such data would allow for improvement of the hydrolysis/solubilization step in the GTLEACH-I.

Furthermore, such data is necessary in order to project the length of time required to complete landfill stabilization and the associated magnitude of leachate strength and migration potential.

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APPENDIX A

DATA ASSESSMENT

INTRODUCTION

The Georgia Institute of Technology research team conducted a review of pertinent available reports of studies on leachate generation and leachate characteristics. The review focused on municipal solid waste (MSW) landfills and pilot-scale studies of leachates generated from MSW and codisposal wastes. Data assessment criteria were investigated to determine the parameters required to model lechate generation in a MSW landfill. It is anticipated that these data assessment criteria may be used to evaluate data from previous landfill investigations or landfill simulation studies. The assessment criteria may also be used for guidance in the preparation of future sampling plans for proposed landfill investigations or landfill simulations.

DATA ASSESSMENT CRITERIA

The generation of leachate from a waste containment structure is a complex process dependent on a large number of variables. Due to the large number of variables and the difficulty in isolating the variables to evaluate potential impact on leachate generation, it is very difficult to quantify and interpret leachate characteristics. However, the ability to predict the life of synthetic or natural liner materials, the potential for leakage, and the impact on the environment requires some quantification of leachate volume and chemical characteristics based on setting, design, waste, and operational variables.

Notwithstanding the difficulty in prediction of leachate characteristics, the available data indicate that the basic processes of MSW landfill stabilization are generally independent of setting, waste, and operational variables. Moreover, the basic processes of biologically-mediated stabilization are similar in lysimeters, pilot-scale landfills and full-scale landfills. However, the extent and rates at which the processes occur and the resultant effect on leachate constituent concentrations is highly dependent on these variables.

The data assessment criteria were developed to provide a mechanism to evaluate the completeness of the data. The data were screened to determine if sufficient data were present to characterize and model the landfill or landfill simulation (i.e., pilot-scale landfill or lysimeter) through the phases of microbial stabilization. The data assessment criteria used in the screening and evaluation of available data are presented and discussed below. These data assessment criteria are divided into five principal areas:

1) Hydrologic Variables;
2) Waste Containment Structure Design Variables;
3) Waste Characteristics;
4) Operational Procedures Affecting Leachate Generation; and
5) Monitoring Data.

Hydrologic Variables

The leaching of liquids through waste materials and the subsequent development of leachate in a landfill is affected by the geologic and hydrologic settings, climate and hydrology as shown in Figure A1. Land use data are also important in the evaluation of potential impacts on receptors.

Site Geologic Model--

The geologic setting is typically described using conceptual regional and site geologic models. The regional geologic model provides a basic understanding of the depositional environment of the soil and rock units in the vicinity of the site. The site geologic model provides a detailed, idealized summary of data obtained during the site investigation. These data typically include boring logs, visual descriptions (micro-logs) of oil specimens and rock cores, and laboratory analyses.

The boring logs may be linked using fence or block diagrams to evaluate the continuity of individual soil or rock units. Of particular interest are the continuity of water bearing units, which may contribute seepage through the liner into the leachate collection system or provide pathways for the release of contaminants, and confining layers, which act as vertical barriers for contaminant migration. The leachate constituents may also interact with the soils, resulting in attentuation, retardation, or fractionation.

The primary properties of interest in the model are those which relate to the transmission of fluid through the layer. These are the soil density, water content, gradation, percent passing the Number 200 sieve, and the hydraulic conductivity or coefficient of permeability. For rock units, the evaluation of properties affecting transmission of fluid is more difficult. An indication of the transmissivity of the rock units may be obtained from core recovery data, in situ evaluation of hydraulic conductivity and regional information on well yield and gradients.

The effectiveness of confining layers and soil barrier layers is typically based on an evaluation of the soil's hydraulic conductivity. However, when soil materials come in contact with high strength leachates, the cation exchange capacity, initial pore water chemistry, state of stress, structure and fabric of the soil may also be important.

The geologic model is required when the landfill is constructed below the original ground surface in such a way that infiltration of ground water may contribute to the quantity of leachate produced.

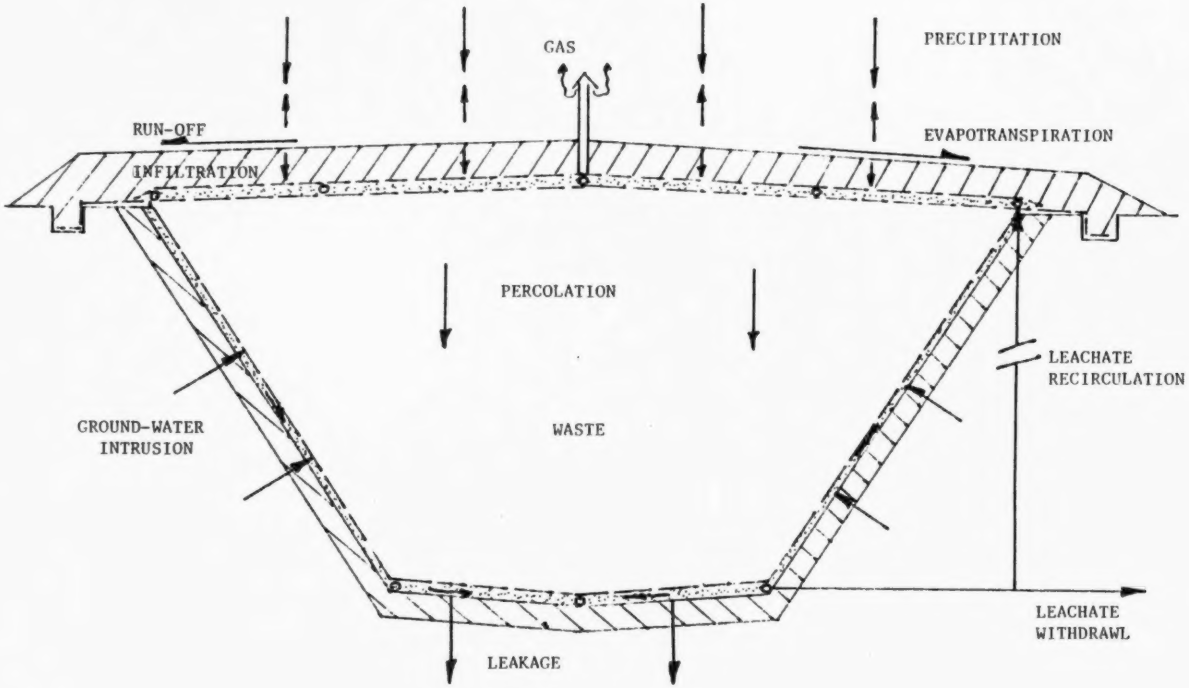
GAS
PRECIPITATION
RUN-OFF
EVAPOTRANSPIRATION
INFILTRATION
PERCOLATION
LEACHATE
RECIRCULATION
GROUND-WATER
INTRUSION
WASTE
LEACHATE
WITHDRAWL
LEAKAGE

FIGURE A1. WATER BALANCE IN A TYPICAL LANDFILL CELL.

Site Hydrogeologic Model--

The regional and site geologic models provide a conceptual description of the occurrence and use of ground water in the vicinity of the site. The regional hydrogeologic model is a companion of the regional geologic model which identifies the major aquifers in the region. The model provides descriptions of the direction of ground water flow, velocity, confinement, and basic water quality.

The site hydrogeologic model describes the occurrence of ground water in the soil and rock units at the site. The water bearing units are typically classified as perched water zones, ground water table zones, or unconfined, leaky or confined aquifers. Data from slug tests or pump tests conducted at the site are used to evaluate the in situ hydraulic conductivities, transmissivities, and storativities of the individual units or layers. Water level data are typically obtained from monitoring wells at the site to evaluate the potentiometric surface for each aquifer as a function of time. Ground water specimens are also obtained for baseline chemical analyses. These analyses typically include all the parameters to be monitored in the monitoring system. The site hydrogeologic model typically focuses on those water bearing units which are in the same horizon as the landfill and the water bearing unit immediately below the landfill.

Ground water use data may also be important in the evaluation of potential impacts on receptors. These data typically include the locations, depths and withdrawal rates of wells in the vicinity of the site, and the locations and types of other potential receptors such as environmentally sensitive areas, basins, lakes and rivers.

Since ground water may contribute to the available water driving the development of the leachate at a landfill, it is necessary to evaluate the regional and site hydrology and to assess their impact on a specified site. The site hydrogeologic model is required when the waste containment facility is constructed below the original ground surface in such a way that infiltration of ground water may contribute to the quantity of leachate produced.

Climatic Data--

The infiltration and percolation of rainfall into landfills provides the major driving force behind the generation of leachate. Therefore, climatic data affecting the quantity of water percolating into a landfill are essential in the evaluation of leachate quantities and characteristics. These climatic data include daily rainfall, solar index, plant uptake, daily temperature, relative humidity and wind speed over time. The climatic data are required whenever the waste containment structure is constructed in such a way that it is open to the elements.

Site Hydrology--

The management of surface water at the site has a major impact on the quantity of water which may enter a landfill. A site plan which details the surface water management strategy should be reviewed to evaluate the potential for surface water run-on. In addition, it is important to obtain data relating to the run-off from the final soil cover. These data would include the type of vegetative cover, the type of soil and the nature of the cover.

A description of surface features such as streams, rivers, lakes and basins should also be provided. This information is typically used in the site hydrogeologic model. The hydrologic data are required whenever there is a possibility of surface run-on affecting the quantity of leachate produced in the landfill.

Waste Containment Facility Design Variables

Type of Landfill--

The type of landfill and its purpose have a major impact on the type, quantity, and rate of generation of constituent concentrations of the leachate. The evaluation focuses on three types of landfills: hazardous waste landfills permitted under RCRA; codisposal landfills which contain both municipal refuse and quantities of hazardous or industrial waste; and, MSW landfills which contain primarily municipal refuse. The design variables and model evaluations described in this report are primarily related to the evaluation of leachate characteristics at MSW landfills. However, the approach and the data requirements are very similar for codisposal and hazardous waste landfills as well as pilot-scale or simulated landfills and lysimeters.

Cell Geometry--

Most landfills are traditionally separated into a series of cells which reduce the initial volumes of water to be handled in a leachate collection system. The cells may be excavated and constructed below the existing ground surface or embankments may be constructed and the cell installed above the ground surface. The configuration of the cell is primarily a function of the local site geology, hydrogeology (depth of the water table), and waste management factors (e.g., site access, size of the facility).

The primary cell geometry variables affecting the leachate are the length, width and depth of the cells, as these variables are related to the waste quantity, and the contact time of the water percolating through the landfill. Side slope data, embankment design data, cut and fill areas and cell locations should also be identified so that the long-term stability of the slopes, drainage capacity of the leachate collection system, and locations of the leachate collection points can be evaluated or identified.

For lysimeters, the basic cell geometry, method of construction, quantity of waste, and treatment of boundaries is required to evaluate the contact time of water percolating through the landfill. It is also important to assess the potential for liquid or gas leakage from the lysimeter and to evaluate potential interaction between the waste and the structure sidewalls.

Liner Design (Soil)--

The design of the soil liner is important for the evaluation of liner/leachate compatibility and the analysis of potential migration from the landfill. The primary variables affecting the performance of the liner are the soil type and classification, the layer thickness, the density and water content, the soil gradation (and hydrometer analysis), the liquid and plastic limits, and the hydraulic conductivity. The field quality control (QC) data are also important in the evaluation of the continuity of the clay liner. Where synthetic liners are used in conjunction with a soil layer to form a composite liner, it is also important to assess the effects of interaction between the synthetic liner and soil. Soil liner design information is not typically required for lysimeters.

Liner Design (Synthetic)--

Synthetic liner design data are required whenever synthetic liner materials are used in the construction of the waste containment structure. The effectiveness of a synthetic liner is primarily a function of the polymer type and density, a method of seaming, the liner thickness, stress/strain behavior, and the installation methodology and quality control. In landfills where a leak detection layer is installed, the performance of the liner may be evaluated directly.

The chemical compatibility of the synthetic liners is typically evaluated using the EPA 9090 evaluation. In the absence of data from the 9090 test, liner compatibility may be evaluated for specific waste streams based on general polymer compatibility data provided by the manufacturer.

Leachate Collection, Leak Detection and Underdrain Layers--

The leachate collection system design information is required for all waste containment structures. The design of the leachate collection, leak detection and underdrain layers affects the quantity of water which is collected in the leachate collection system, the contact time between the liquid and the waste, and the time required to reach field capacity in the landfill. The primary variables affecting the performance of the drainage layers are the flow capacity and the thickness of the drain. For aggregate drains, the type of aggregate, grain size distribution, gradient, and hydraulic conductivity control the flow capacity of the drain. For synthetic drainage materials, the type of drain, the polymer type and density, gradient, and the transmissivity under load control the flow capacity of the drain. Flow capacity reduction due to microbial growth in the drainage layers should be assessed.

When geotextiles are used for filtration or separation, the flow through the geotextile into the drainage layer becomes important. The primary variables affecting flow normal to the geotextile are the type, thickness, porosity and hydraulic conductivity ratio of the geotextile. Flow reduction due to microbial growth in the drainage layers should be addressed in the evaluation of leachate management for the landfill.

Final Cover--

The design of the final cover system has a major impact on the generation of leachate. Since the driving force behind the generation of leachate is infiltration and percolation, the design of the cover system, which limits the amount of water available for percolation, is critical to the evaluation of leachate generation as a function of time. The primary variables required to evaluate infiltration through the cover are the final cover profile, the soil types, the average densities and water contents of each soil layer at placement, the liquid and plastic limits, grain size distribution and vertical hydraulic conductivity of the soil layer, and the type of vegetative cover. In addition to these parameters, the overall surface water management program should be investigated. This would include the design and layout of the surface water management facilities on the cover.

Since the quantity of water which percolates through the landfill is much greater prior to closure, it is important to determine when the final cover is placed on each cell of the landfill. The cover design information is required for all waste containment structures which are constructed and operated out of doors.

Daily Cover--

The placement of daily or intermediate cover affects the migration of rain water and leachate through the landfill. The primary variables required to assess the impact on migration are the type of soil, thickness and locations of the soil layers, the compactive effort and the hydraulic conductivity. In many landfills, the daily cover is removed in some areas, particularly adjacent to the cover on a slope, to promote vertical percolation. It is important to record the locations of all areas where the daily cover is removed.

Daily cover information is required whenever intermediate soil layers are placed between waste layers in the landfill.

Gas Collection System--

Where gas collection systems have been installed, preferential paths for the migration of leachate may have been created. Therefore, the design of the gas collection system, its layout and the probability of leachate migration through or around the gas collection system should be assessed.

On-Site Surface Structures or Appurtences

Surface impoundments and tanks are often used to temporarily store leachate or storm water prior to collection or treatment. These impoundments and tanks may leak, creating a ground-water mound near portions of the landfill. The impact of these facilities on the potentiometric surface and groundwater quality should be assessed. The primary variables required to evaluate the potential for leakage from impoundments or ponds are the size of the pond, the depth of fluid in the pond, the pond location, and the liner type, thickness, and hydraulic conductivity.

Shallow groundwater monitoring wells in the vicinity of the impoundment, tanks or ponds should be evaluated to determine if leakage has occurred. Water level readings from the wells should be recorded on a regular basis to assess the impact of potential leakage on the potentiometric surface.

These data are only required when the waste containment facility is constructed below the original ground surface and there is a reasonable possibility that flow from the surface impoundments or tanks may contribute to the generation of leachate.

Waste Characteristics

Waste characteristics have a major impact on leachate composition and constituent concentrations, stabilization processes, time required for each phase of stabilization to occur, and the rate of migration of leachate through the waste. Due to the heterogeneity of typical municipal wastes, sufficient analyses are required to characterize the chemical composition of the waste. Such analyses would likely be expensive, and would be applicable only to the wastes at the landfill where the analyses are performed. In the absence of such chemical composition data, it has been traditional to classify wastes according to their origin.

Composition of Solid Wastes--

Pohland and Harper (1985) classified solid wastes into five major source categories including residential, agricultural, commercial, municipal, and industrial. Waste originating from each source category are typically dominated by a limited number of constituents. For example, the major constituents of residential waste are rubbish, food and garden wastes, paper, plastics, glass and ash. Household hazardous wastes are also included in the residential waste category, but are generally relatively minor compared to the total volume of the major constituents. Agricultural wastes consist primarily of crop and animal wastes, food wastes, rubbish, and chemicals such as pesticides, herbicides and insecticides. The major constituents of commercial waste are rubbish, food waste,construction/demolition debris, ash and chemical waste residues. Municipal wastes often consist of combinations of the other categories with an abundance from the residential and commercial sectors. Industrial waste streams are a function of the industry from which they originate. These wastes typically consist of biological and chemical sludges, chemicals, rubbish, ash, construction/demolition debris and chemical waste residues.

Food wastes, rubbish, garden wastes, crop and animal residues and sludges would contribute organic components to solid wastes. These organic compounds often serve as growth substrates and nutrients for the microbial populations present in a landfill as stabilization proceeds. These wastes often also contribute moisture for leachate formation and a biological inoculum for biological activity leading to conversion products of volatile organic acids and gas. Chemical sludges, chemical waste residues, chemicals and industrial and municipal sludges typically contribute inorganic as well as more complex organic constituents which provide sources of hazardous or toxic heavy metals and organic chemicals. The volume of hazardous or toxic heavy metals and organic chemicals is generally small in an MSW landfill, but could constitute a potentially large source of hazardous constituents in codisposal or hazardous waste landfills.

The character of leachate is largely a function of the characteristics of the waste in the landfill. Where significant quantities of heavy metals or hazardous organic constituents are included in the waste, these constituents should be expected to appear in the leachate depending on the phase of stabilization. Where waste materials are disposed which lead to very high (>12) or very low (<2) pH, or when highly polar organic chemicals are present, the interaction of the leachate with the liner should be assessed to evaluate potential changes in liner properties which may lead to increased rates of leakage or seepage from the landfill.

Relatively high or low pH conditions or the presence of toxic inorganic and/or organic chemicals may inhibit the microbially mediated processes of landfill stabilization. However, research indicates that disposal of appropriate proportions of industrial and municipal wastes may be accomplished with no adverse effects on the stabilization processes (Houle, et al., 1977; Barber, 1981; Pohland, et al., 1981; and Walsh, et al., 1983). In some cases, the moisture from industrial sludges actually accelerates the stabilization process (Walsh, et al., 1983) when inhibition of the microbial processes occurs, the stabilization processes typically progress at a reduced rate (if at all) as the leachate tends to become more concentrated. This may increase the potential for migration of certain inorganic and organic constituents which become more mobile under acid or low pH conditions.

Operational Procedures Affecting Leachate Generation

Two landfills, in similar regions of the country, with similar geologic and hydrogeologic settings and identical waste streams can have vastly different leachates resulting from differences in the operational procedure for the landfills. The primary operational variables affecting leachate generation are the waste placement conditions and methods, the leachate collection procedures and system design, and possible leachate recirculation procedures.

Waste Placement--

The method of waste placement affects both the density and hydraulic conductivity of the waste after compaction. The vertical hydraulic conductivity of the waste is primarily a function of the structure, density, water or liquid content, and particle size distribution of the waste. The relative compaction may be estimated based on the type of waste, the initial moisture content, the method of placement (end dump, spreading, etc.), lift thickness prior to compaction, and type of compactor used for compaction. Very little is known about the vertical hydraulic conductivity in saturated or partially saturated waste materials. The vertical saturated hydraulic conductivity may vary from greater than $1x10^{-1}$ cm/sec for poorly compacted construction debris to less than $1x10^{-6}$ cm/sec for some sludge materials. In addition, density changes, changes in moisture content and differential settlement in the waste occur as a function of time during the stabilization process. Therefore, it is likely that the vertical hydraulic conductivity of the waste also changes with time.

The cell dimensions and dates of waste and final cover placement are also important for the evaluation of vertical percolation. Information regarding the placement and compaction of the waste materials is required for all waste containment structures.

Leachate Management--

The management of leachate in the landfill is extremely important for the evaluation of leachate generation and chemical characteristics. The primary leachate management variables affecting leachate composition are the detention times, locations of leachate detention (e.g., landfill, sump, surface impoundment) and locations of leachate removal.

In addition to the leachate management information, data concerning the removal of leachate from the landfill are important. These data include the dates and volumes of leachate removed, the locations from which leachate was removed, the depths of leachate in each cell, and the first date leachate was detected in the leachate collection system.

Field monitoring data, such as the chemical analysis of the leachate for the indicator parameters discussed under "Leachate Monitoring," and visual descriptions such as the color, turbidity and odor may also be useful.

The leachate management information is essential to evaluate the rate of generation and quantity of leachate and to evaluate the distribution of water as a function of time and space in the waste. Since the characteristics of the leachate are controlled by these variables, leachate management information is required for all landfills.

Leachate Recirculation--

Landfill stabilization processes tend to be accelerated when the moisture content remains at or near the field capacity of the waste. Leachate recirculation increases the moisture content of the waste, more evenly distributes nutrients and enhances biomass/substrate contact, and maintains the waste at or near the field capacity, thus accelerating the stabilization processes.

The primary variables which must be considered to evaluate the impact of leachate recirculation on the stabilization processes are the volumes, and rates and extent of application, recirculation schedules, and the chemical character of the recycled leachate. Gas generation data are also important to the overall evaluation.

Gas Management--

While the gas management system does not have an immediate impact on leachate characteristics, monitoring of the gas collection or gas venting system can reveal valuable information regarding the condition of stabilization in the landfill. Gas production occurs mainly during the later phases of landfill stabilization and typically indicates that the stabilization process is in an advanced state of progression. The primary variables which should be evaluated are the date of first measurable gas generation, the dates of gas measurements and the gas generation rates and quality. In order to model the landfill or lysimeter through the various stages of microbial stabilization the gas generation data are required for all of the waste containment structures.

Monitoring Data

Monitoring data are an essential source of information required to evaluate the maturity of a waste containment facility. Analyses are typically performed on the leachate, ground water, surface water, gas, and wastes. The results of these analyses, combined with information regarding the frequency of monitoring and the sample locations may be used to predict short- and long-term leachate constituent concentrations.

Leachate Monitoring Data--

Monitoring data may prove to be a very valuable source of information in the evaluation of leachate characteristics. As discussed by Pohland and Harper (1985), there are several indicator parameters which may be used to describe the presence, intensity and longevity of the phases of landfill stabilization. When applied to leachate, these indicator parameters provide insight into the mobility of certain organic and inorganic constituents. Table A1 provides a summary of parameters which provide useful information regarding the stabilization processes in a landfill.

As suggested in Table A1, there are physical, chemical or biological indicator parameters which provide information required to evaluate the interrelationships of the landfill stabilization processes. As an example, pH and ORP are physical-chemical paramaters used to indicate acid-base and oxidation-reduction conditions critical to the evaluation of the acidogenesis and methanogenesis processes. The chemical and biological parameters of COD and BOD_5, respectively, are used to evaluate relative biodegradability. The chemical parameters, nitrogen and phosphorus, are used to evaluate nutrient sufficiency and condition (aerobic/anaerobic) of a particular phase.

In addition to the parameters discussed above, the following additional parameters may be used to assess stabilization in terms of their significance as indicated below:

Parameter	Process
Total Alkalinity	Buffer Capacity
Heavy Metals	Potential Inhibition
Specific Conductance	Ionic Strength/Activity
Chlorides	Migration Potential
Bacteria & Viruses	Health Hazards
Nitrates & Sulfates	Oxidizing Potential or Condition

As illustrated in Figure A2, the concentrations of selected indicator parameters vary during each phase of landfill stabilization. Review of available data indicate that while some contradictions occur, indicator parameter concentrations for leachates from both landfills and lysimeters, whether with or without microbial inhibition or leachate recycle, will be similar but a varying times and perhaps for different reasons.

For instance, the magnitude of leachate concentrations during each phase of stabilization is strongly affected by dilution. However, dilution does not affect the total mass of leached constituents in space and time. Unfortunately, the effects of dilution and the distribution of indicator parameter concentrations with time are often poorly recorded, making it difficult to assess which phase of stabilization a landfill is currently undergoing. In addition, there may be discrete compartment of pockets of activity where the stabilization processes are occuring at different stages

TABLE A1. LANDFILL LEACHATE AND GAS CONSTITUENT CONCENTRATION RANGES ENCOUNTERED IN THE LITERATURE*AND THEIR RELATIVE SIGNIFICANCE TO THE DEGREE OF LANDFILL STABILIZATION.

Leachate or Gas Constituent	Phase of Biological Stabilization				Overall Range (All Phases)
	Transition Phase	Acid Formation Phase	Methane Fermentation Phase	Final Maturation Phase	
Biochemical Oxygen Demand (BOD_5)	100-10,900 Influence of dilution and aerobic solubilization of waste organics	1,000-57,700 Accumulation of biodegradable organic acids due to methanogenic lag	600-3,400 Conversion of biodegradable organics to gaseous end products (CH_4 and CO_2)	4-120 Influence of high-molecular weight organic residuals (humics, fulvics)	4-57,700
Chemical Oxygen Demand (COD)	480-18,000 Trending in a similar fashion to BOD_5	1,500-71,100 Trending in a similar fashion to BOD_5	580-9,760 Trending in a similar fashion to BOD_5	31-900 Higher influence of residual organics than in BOD_5 assay.	31-71,700
Total Organic Carbon (TOC), mg/l	100-3,000 Beginning to appear as a result of aerobic solubilization	500-27,700 Increasing rapidly; accumulation due to methanogenic lag	300-2,230 Conversion of volatile acids to methane; decrease in aqueous carbon	70-260 Influence of higher molecular weight organics (humics, fulvics)	70-27,700
Total Volatile Acids (TVA), mg/l as Acetic Acid	100-3,000 Or just beginning to appear as a result of solubilization	3,000-18,800 Solubilization of organic polymers to monomers; beta-oxidation to volatile acids	250-4,000 Conversion of fatty acids to fatty acid; fermentation of acetic acid to methane	Essentially absent - methanogenic system undersaturated	0-18,800
BOD_5/COD Ratio	0.23-0.87 Increasing biodegradability of organics due to solubilization	0.4-0.8 High biodegradability	0.17-0.64 Decreasing biodegradability due to methanation	0.02-0.13 Low degree of biodegradability	0.02-0.87
COD/TOC Ratio	4.3-4.8 Low oxidation state or organics	2.1-3.4 Low to moderate oxidation state of organics	2.0-3.0 Moderate to high oxidation of organics	0.4-2.0	0.4-4.8
Total Kjeldahl Nitrogen (TKN) mg/l	180-860	14-1,970 May be low due to microbial assimilation of nitrogenous compounds	25-82 Low due to microbial assimilation of nitrogenous compounds	7-490	7-1,970
Nitrate Nitrogen (NO_3^--N), mg/l	0.1-5.1 Increasing due to oxidation of ammonia	0.05-19 Decreasing due to reduction to ammonia or N_2 gas	Absent Complete conversion to ammonia or N_2 gas	0.5-0.6	0-51
Ammonia Nitrogen (NH_3-N) mg/l	120-125	2-1,030 Increasing due to nitrate reduction and protein breakdown	6-430 Decreasing due to biological assimilation	6-430	2-1,030
NH_3/TKN Ratio	0.1-0.9	0-0.98 Protein breakdown; biological assimilation	0.1-0.84	0.5-0.97	0.1
Total Phosphate ($PO_4^{=}$-P), mg/l	0.6-1.7	0.2-120 Biological assimilation and metal complexation	0.7-14 Low due to biological assimilation	0.2-14	0.2-120

TABLE A1. LANDFILL LEACHATE AND GAS CONSTITUENT CONCENTRATION RANGES ENCOUNTERED IN THE LITERATURE AND THEIR RELATIVE SIGNIFICANCE TO THE DEGREE OF LANDFILL STABILIZATION (continued).

Total Alkalinity, mg/l as $CaCO_3$	200-2,500	140-9,650 Increasing due to volatile acid formation and bicarbonate dissolution	760-5,050 Decreasing due to volatile acid removal	200-3,520	140-9,650
Solids (TS), mg/l	2,450-2,050	4,120-55,300 Increasing due to solubilization or organics and mobilization of metals	2,090-6,410	1,460-4,640	1,460-55,300
pH	6.7	4.7-7.7 Low due to volatile acid accumulation	6.3-8.8 Increasing due to volatile acid removal and bicarbonate dissolution	7.1-8.8	4.7-8.8
Oxidation-Reduction Potential (ORP), mV	+40 to +80	+80 to -240 Decreasing due to the depletion of oxygen	-70 to -240	+97 to +163	-240 to +16
Copper, mg/l	0.085-0.39	0.005-2.2	0.03-0.18 Decreasing (complexation)	0.02-0.56	0.005-2.2
Iron, mg/l	68-312	90-2,200	115-336 Decreasing (complexation)	4-20	4-2,200
Lead, mg/l	0.001-0.004	0.01-1.44	0.01-0.1 Decreasing (complexation)	0.01-0.1	0.001-1.44
Magnesium, mg/l	66-96	3-1,140	81-505 Decreasing (complexation)	81-190	3-1,140
Manganese mg/l	0.6	0.6-41	0.6 Decreasing (complexation)	0.6	0.6-41
Nickel, mg/l	0.02-1.55	0.03-79	0.01-1.0 Decreasing (complexation)	0.07	0.02-79
Potassium, mg/l	35-2,300	35-2,300	35-2,300	35,2,300	35-2,300
Sodium, mg/l	20,7,600				20-7,600
Zinc, mg/l	0.06-21	0.65-220	0.4-6.0	0.4	0.06-220
Total Coliform, CFU/100 ml	10^0 to 10^5	10^0 to 10^5	Essentially absent	Absent	$0-10^5$
Fecal Coliform, CFU/100 ml	10^0 to 10^5	10^0 to 10^5	Essentially absent	Absent	$0-10^5$
Fecal Streptococci, CFU/100 ml	10^0 to 10^6	10^0 to 10^6	Essentially absent	Absent	$0-10^6$

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TABLE A1. LANDFILL LEACHATE AND GAS CONSTITUENT CONCENTRATION RANGES ENCOUNTERED IN THE LITERATURE AND THEIR RELATIVE SIGNIFICANCE TO THE DEGREE OF LANDFILL STABILIZATION (continued).

Viruses, PFU/100 ml	--	Essentially absent	Essentially absent	Essentially absent	Absent
Conductivity, μmhos/cm	2,450-3,310	1,600-17,100 Increasing due to mobilization of metals	2,900-7,700 Decreasing due to metals complexation with sulfides	1,400-4,500	1,400-17,100
Chloride (Cl^-), mg/l	30-5,000 Biologically stable; good indicator of washout	30-5,000 Stable; good hydraulic tracer	30-5,000 Stable; good hydraulic tracer	30-5,000 Stable; good hydraulic tracer	30-5,000
Sulfate ($SO_4^=$), mg/l	10-458 Increasing due to aerobic oxidation	10-3,240 Increasing initially due to aerobic solubilization then decreasing as anaerobic conditions are estabilished	Absent Complete conversion to sulfides	5-40 Reappearing due to aerobic oxidation	
Sulfide ($S^=$), mg/l	Essentially absent	0-818 Beginning to appear and increasing due to sulfate reduction under anaerobic conditions	0.9 Low due to heavy metal precipitation	Absent	
Cadmium, mg/l	190-490	70-3,900	76-490 Decreasing due to complexation and precipitation	76-254	70-3,900
Chromium, mg/l	0.023-0.28	0.06-18	0.05 Decreasing due to complexation, precipitation with sulfides	0.05	0.02-18
Methane, %	Essentially absent (aerobic metabolism)	Very low (<1%); Transition to anaerobic metabolism	30-60 Suitable for energy recovery	0-<10 Decreasing due to substrate limitation and reversion to aerobic metabolism	0-60
Carbon Dioxide, %	0-10 Product of aerobic decomposition of organics	10-30 Increasing due to waste decomposition	30-60 Decreasing to <5% as methanogenesis increases	<40 Aerobic metabolism	0-60
Nitrogen Gas, %	70-80 Influence of trapped air	60-80 Decreasing due to dilution with CO_2	<20 Artefact of trapped air; denitrification	>20 Increasing due to introduction of air	<20-80
Oxygen, %	20 Influence of trapped air	0-5 Decreasing due to aerobic utilization; shift towards anaerobic metabolism	0-5 Disappearing as methanogenesis increases	>5 Increasing due to introduction of air	0-20
Hydrogen, %	Essentially absent in the presence of oxygen	0-2 Beginning to appear as oxygen is depleted; accumulates until methanogenesis occurs	<0.1 Maintained at low levels by methanogenesis; difficult to measure	0-2 Essentially absent	

[*] Ranges of constituent concentrations were collected from the references and data presented in the Appendices.

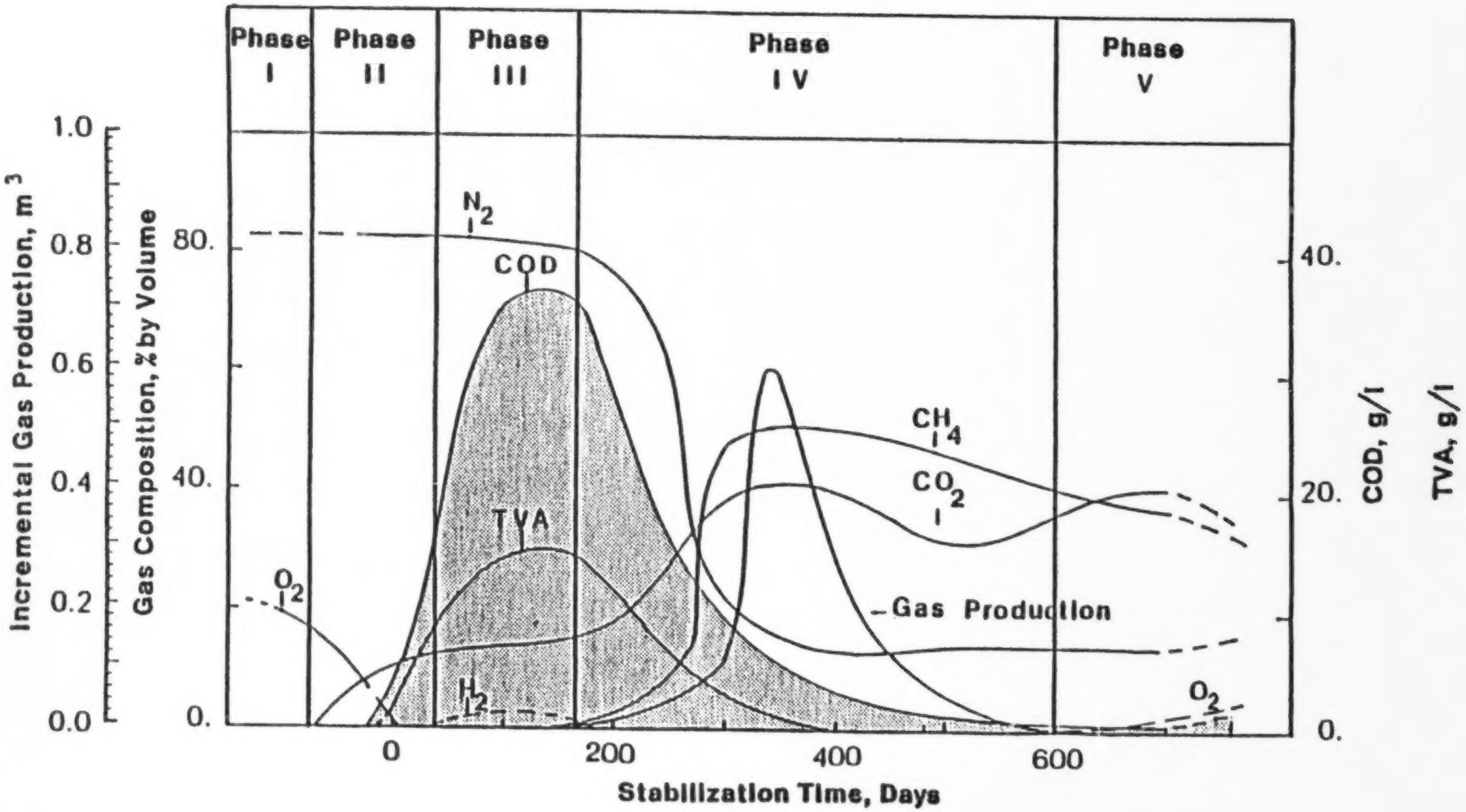

FIGURE A2. CHANGES IN SELECTED INDICATOR PARAMETERS DURING THE PHASES OF LANDFILL STABILIZATION (After Pohland, 1980 and Pohland, et al., 1983).

and rates, often making it difficult to distinguish between phases. Therefore, in order to properly assess the impact of dilution, indicator parameters should be frequently evaluated on a mass basis. Such an evaluation requires careful monitoring of leachate volumes and evaluation of infiltration, percolation and recycle quantities and rates. Where flow data are not recorded, it may not be possible to distinguish between the intensities of various phases of landfill stabilization or those consequenced by microbial mediation or dilution or both.

Ground-water and Surface Water Monitoring Data--

Ground-water and surface water monitoring are required to provide an early warning system in the event of leakage from a waste containment facility. Ground-water and surface water monitoring data are also useful for the evaluation of seasonal variations in the potentiometric surface, which may affect the direction and velocity of ground-water flow.

The ground-water and surface water elevations should be obtained on a regular basis and the potentiometric surface plotted as a function of time. Samples should be obtained and analyzed for the same constituents and indicator parameters monitored in the leachate. These data are required for all waste containment structures where ground water may enter the leachate collection system.

Gas Monitoring Data--

Gas monitoring data are required for the evaluation of landfill maturation. Since methane production is pronounced during the methane fermentation phase of landfill stabilization, careful monitoring of gas generation rates at the landfill is required to indicate when the landfill has completed the active phases of stabilization and has reached the relative dormancy of the final maturation phase. These data are required for all waste containment facilities.

Waste Monitoring Data--

Many landfills are segregated for various wastes, leading to variations in leachate characteristics from cell to cell or even point to point within a cell. Data regarding the segregation of different types of wastes should be scrutinized to assess potential impacts on leachate characteristics.

In addition, the physical characteristics of the waste such as density, porosity, specific surface area and hydraulic conductivity change as a function of time and depth. In order to model flow through a landfill, it may be necessary to measure the variation in the parameters as a function of time.

General Perspective

The evaluation of leachate volumes and chemical characteristics is extremely complex and dependent on a large number of variables. The impact of many of these variables on leachate generation has often been overlooked, resulting in wide variation in interpretations and prediction of actual performance.

An important factor in the generation of leachate is the quantity of water entering and the rate at which water percolates though the waste materials. The hydrologic and waste containment facility design variables determine the rate of leachate generation progress of microbially mediated waste stabilization and the leachate and gas characteristics with time. Figure 1 shows typical contributions and withdrawals of water at a landfill cell. The hydrologic and landfill design variables may be used to perform a water balance for each landfill cell. The water balance may be used to estimate the quantity of water percolating into the waste and the total volume and rate of leachate produced.

Since the microbially-mediated processes of landfill stabilization are controlled primarily by the waste type, the availability of nutrients and the moisture content of the waste, waste characterization and waste placement data are vital. These data are used to estimate the initial substrate availability and vertical hydraulic conductivity of the waste. The hydraulic conductivity, combined with information regarding the waste porosity may then be used to evaluate the rate at which field capacity would be reached, the potential magnitude of short-circuiting and the rate of leachate production. The waste characteristics and waste constituent concentrations may also be used to predict leachate constituent concentrations as a function of time and to assess the magnitude and impact of possible retardation or inhibition of stabilization.

APPENDIX B

PROGRAM GTLEACH-I

```
C
C
C THIS PROGRAM IS DESIGNED TO SIMULATE THE FOLLOWING CHARACTERISTICS
C OF LEACHATE EMANATING FROM SANITARY LANDFILLS DURING STABILIZATION
C --VOLUMETRIC FLOW RATE, CHEMICL OXYGEN DEMAND, VOLATILE ACID
C CONCENTRATION, AND PH.
C
C
      EXTERNAL FCN,FCNJ
      COMMON YY(20,6),NDAYS,NCSTR,VOL,TVOL,ZZ(20,6)
      COMMON XKH,YA,QA,XKA,XKDA,YM,QM,XKM,XKDM,F,LAG
      COMMON SAO,SHO,XMASSO,XAO,XMO,IRECYC,I,J,K,IOUT,GASMET
      COMMON N,METH,MITER, INDEX,IWK(5),IER
      COMMON Y(5),WK(86),X,TOL,XEND,H
C
C
      OPEN (8,FILE='PARAM.DAT',STATUS='OLD')
      OPEN(7,FILE='RESULT.DAT')
C
C
C
      CALL INPUT
C
C INITIALIZES VALUES FOR ALL CSTRS BASED ON INPUT DATA
C
      DO 25 JJ=1,NCSTR
      YY(JJ,1)=XMASSO
      YY(JJ,2)=XAO
      YY(JJ,3)=0.0
      YY(JJ,4)=SHO
      YY(JJ,5)=SAO
25    CONTINUE
C
      WRITE(7,33)
33    FORMAT(2X,'TIME',4X,'XMASS',6X,'XA',8X,'XM',8X,'SH',8X,'SA',7X,
     1'COD',5X,'Methane')
C
      IF (IOUT.EQ.0) THEN
C
      WRITE(7,44)
44    FORMAT(1X,'(days)',2X,'(mg/1)',4X,'(mg/1)',4X,'(mg/1)',4X,
     1'(mg/1)',4X,'(mg/1)',4X,'(mg/1)',4X,'(g/day)')
C
```

```
      ELSEIF (IOUT.EQ.1) THEN
C
      WRITE(7,55)
55    FORMAT(1X,'(days)',3X,'(kg/m3)',4X,'(mg/l)',6X,'(mg/l)',6X,
     1'(grams)',4X,'(grams)',4X,'(grams)',5X'(g/day)')
C
      ELSE
      ENDIF
C
C
      II=1
C
      DO 175 I-1,NDAYS
C
      XEND-FLOAT(I)
C
C
C AT SPECIFIED TIME=LAG, INITIALIZES CONCENTRATION OF METHANOGENS
C
      IF (I.EQ.LAG) THEN
      DO 65 JJ = 1, NCSTR
      YY(JJ,3) = XMO
65    CONTINUE
      ELSE
      ENDIF
C
C FOR FIRST 50 DAYS INCREMENTS XMASSO
C
      IF (I.LT.51) THEN
      DO 66 JJ=1,NCSTR
      YY(JJ,1)=YY(JJ,1) + 2.2E+05/TVOL
66    CONTINUE
      ELSE
      ENDIF
C
C
C AT TIME=200 DAYS INCREASES XMASSO TO SIMULATE NEW SUBSTRATE
C
      IF (I.EQ.200) THEN
      DO 77 JJ=1,NCSTR
      YY(JJ,1)= (1.0E+08)/TVOL
77    CONTINUE
      ELSE
      ENDIF
C
C
C
C
C AT EACH NEW TIME STEP THE INFLUENT VALUES SAO AND SHO ARE
C SET FOR THE FIRST CSTR DEPENDING ON RECYCLE STATUS.
C
      IF (IRECYC.EQ.0) THEN
```

```
        SHO=0.0
        SAO=0.0
        ELSEIF(IRECYC.EQ.1)THEN
        SHO=Y(4)
        SAO=Y(5)
        ELSE
        ENDIF
C
                     DO 150 J=1,NCSTR
C
C SETS INITIAL VALUES OF Y'S FOR CSTR EQUAL TO THOSE FOR THE SAME
C CSTR AT THE PREVIOUS TIME STEP AS STORED IN YY.
C
C
                            DO 50 K=1,N
                            Y(K)=YY(J,K)
50                          CONTINUE
C
C INITIALIZES VALUE OF TIME FOR EACH CSTR CALCULATION
C
                     X=XEND-1.0
C
C CALLS DGEAR TO SOLVE DIFFERENTIAL EQUATIONS FOR CSTR
C
                     CALL DGEAR (N,FCN,FCNJ,X,H,Y,XEND,TOL,METH,MITER,
     1INDEX,IWK,WK,IER)
        IF(IER.GT.128) GO TO 999
C SETS INFLUENT VALUES FOR NEXT CSTR STEP
C
                            SHO=YY(J,4)
                            SAO=YY(J,5)
C STORES SOLUTIONS IN MATRIX YY FOR USE AS INITIAL VALUES IN
C NEXT TIME STEP
C
                            DO 100 K = 1,N
                            YY(J,K) = Y(K)
100                         CONTINUE
C
                            INDEX=1
C
150                 CONTINUE
C
C CALLS OUTPUT TO WRITE VALUES OF LAST CSTR TO RESULT.DAT
C
                    IF(I.EQ.II)THEN
                    CALL OUTPUT
                    II=II+5
                    ELSE
                    ENDIF
C
C SEE DGEAR DOCUMENTATION FOR EXPLANATION OF INDEX
C
```

```
175     CONTINUE
        STOP
999     CONTINUE
        STOP
        END
C
C
        SUBROUTINE FCN(N,X,Y,YPRIME)
C
C CONTAINS THE SYSTEM OF DIFFERENTIAL EQUATIONS FOR DGEAR
C
C
        REAL Y(N),YPRIME(N),X
        COMMON YY(20,6),NDAYS,NCSTR,VOL,TVOL,ZZ(20,6)
        COMMON XKH,YA,QA,XKA,XKDA,YM,QM,XKM,XKDM,F,LAG
        COMMON SAO,SHO,XMASSO,XAO,XMO,IRECYC,I,J,K,IOUT,GASMET
C
C
C Y(1)=XMASS=SOLID SUBSTRATE CONCENTRATION
C Y(2)=XA=ACIDOGEN CONCENTRATION
C Y(3)=XM=METHANOGEN CONCENTRATION
C Y(4)=SH=HYDROLYSIS PRODUCT CONCENTRATION
C Y(5)=SA=VOLATILE ACID CONCENTRATION
C
C
        YPRIME(1)=-XKH*Y(1)
        YPRIME(2)=YA*QA*Y(4)*Y(2)/(XKA+Y(4)) - XKDA*Y(2)
        YPRIME(3)=YM*QM*Y(5)*Y(3)/(XKM+Y(5)) - XKDM*Y(3)
        YPRIME(4)=XKH*Y(1) - QA*Y(4)*Y(2)/(XKA+Y(4)) + F*(SHO-Y(4)
     1)/VOL
        YPRIME(5)=Y(4)*Y(2)/(XKA+Y(4)) - QM*Y(5)*Y(3)/(XKM_Y(5) + F*
     1(SAO-Y(5))/VOL
C
        RETURN
        END
C
C
C
        SUBROUTINE FCNJ(N,X,Y,PD)
C
C DUMMY FUNCTION FOR MITER=0
C
        INTEGER N
        REAL Y(N),PD(N,N),X
        RETURN
        END
C
C
C
        SUBROUTINE INPUT
C
        COMMON YY(20,6),NDAYS,NCSTR,VOL,TVOL,ZZ(20,6)
```

```
      COMMON XKH,YA,QA,XKA,XKDA,YM,QM,XKM,XKDM,F,LAG
      COMMON SAO,SHO,XMASSO,XAO,XMO,IRECYC,I,J,K,IOUT,GASMET
      COMMON N,METH,MITER,INDEX,IWK(5),IER
      COMMON Y(5),WK(86),X,TOL,XEND,H
C
      READ (UNIT=8,FMT=*) N,METH,MITER,INDEX,H,TOL
      READ (UNIT=8,FMT=*) F,TVOL,XMASSO,SHO,SAO,XAO,XMO
      READ (UNIT=8,FMT=*) IOUT,LAG
      READ (UNIT=8,FMT=*) XKA,XKDA,YA
      READ (UNIT=8,FMT=*) XKM,XKDM,YM
      READ (UNIT=8,FMT=*) XKH,QA,QM
      READ (UNIT=8,FMT=*) NCSTR,NDAYS,IRECYC
C
      VOL=TVOL/NCSTR
      XMASSO = XMASSO*1.0E+06/TVOL
C
      RETURN
C
      END
C
      SUBROUTINE OUTPUT
C
      COMMON YY(20,6),NDAYS,NCSTR,VOL,TVOL,ZZ(20,6)
      COMMON XKH,YA,QA,XKA,XKDA,YM,QM,XKM,XKDM,F,LAG
      COMMON SAO,SHO,XMASSO,XAO,XMO,IRECYC,I,J,K,IOUT,GASMET
C
C CALCULATES MASS OF METHANE PRODUCED
C
      GASMET=0.0
      DO 195 M=1,NCSTR
      GASMET = GASMET + VOL*0.267*QM*YY(M,5)*YY(M,3)/((XKM+
     1YY(M,5))*100.)
195   CONTINUE
C
      IF (IOUT.EQ.0) THEN
C
      YY(NCSTR,6)=YY(NCSTR,4) + YY(NCSTR,5)*2.5
      WRITE (7,203)I,(YY(NCSTR,M),M=1,6),GASMET
203   FORMAT (1X,I4,2X,E8.3,2X,E8.3,2X,E8.3,2X,E8.3,2X,E8.3,2X,E8.3,
     12X,E8.3)
C
C CONVERTS TO MASS IF IOUT=1
C
      ELSEIF (IOUT.EQ.1) THEN
C
      ZZ(NCSTR,1) = YY(NCSTR,1)
      ZZ(NCSTR,2) = YY(NCSTR,2)
      ZZ(NCSTR,3) = YY(NCSTR,3)
      ZZ(NCSTR,4) = YY(NCSTR,4)*F/100.
      ZZ(NCSTR,5) = YY(NCSTR,5)*F/100.
      ZZ(NCSTR,6) = ZZ(NCSTR,4) + ZZ(NCSTR,5)*2.5
C
```

```
      WRITE (7,205)I,(ZZ(NCSTR,M),M=1,6),GASMET
205   FORMAT (1X,I4,2X,E8.3,2X,E8.3,2X,E8.3,2X,E8.3,2X,E8.3,2X,E8.3,
     12X,E8.3)
C
      ELSE
      ENDIF
C
      RETURN
      END




C     CCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCC
C     C                                                C
C     C        THE PROGRAM SOLVE FLUID VELOCITY        C
C     C                     OF                         C
C     C          GROUNDWATER (SOIL SATURATED)          C
C     CCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCCC
C     PROGRAM SAT
      DIMENSION XK(10),XL(10),HEAD(100),TIME(100),TR(100),QDUMP(100)
      CHARACTER FIN*8,FOUT*8
      WRITE(*,6)
    6 FORMAT(' INPUT NAME OF INPUT FILE (EX:XYZ.DAT) -',\)
      READ(*,8)FIN
    8 FORMAT(A8)
      WRITE(*,7)
    7 FORMAT(' INPUT NAME OF OUTPUT FILE (EX:XYZ.OUT) -',\)
      READ(*,8)FOUT
      OPEN(5,FILE=FIN,STATUS='OLD')
      OPEN(6,FILE=FOUT,STATUE='NEW')
C
C
C     INPUT DATA:
C
C           NLAYER : NO. OF LAYERS
C           NCYCLE : NO. OF CYCLES
C           XK    : HYDRAULIC CONDUCTIVITY OF SOIL LAYER (CM/SEC)
C           XKSS : HYDRAULIC CONDUCTIVITY OF SHORT CIRCUITING (CM/SEC)
C           EN    : EFFECTIVE POROSITY (-) (0.01 to 0.1)
C           TFSS  : ELAPSED TIME OF SHORT CIRCUITING (10 TO 100 DAYS)
C           XL    : THICKNESS OF SOIL LAYER (FT.)
C           QDUMP : QUANTITY OF WATER DUMPED IN LANDFILL (CU-FT.)
C           AREA  : AREA OF LANDFILL (SQ-FT.)
C           TIME  : ELAPSED TIME OF EACH DUMPED WATER (DAYS)
C
C
      READ(5,*)NLAYER,AREA,XKSS,EN
      DO 10 I=1,NLAYER
         READ(5,*)XK(I),XL(I)
         XL(I)=XL(I)*12*2.54
      10 CONTINUE
          READ(5,*)NCYCLE,TFSS
          TFSS=TFSS*3600*24
          DO 20 I=1,NCYCLE
             READ(5,*)QDUMP(I),TIME(I)
             HEAD(I)=QDUMP(I)/AREA
             HEAD(I)=HEAD(I)*12*2.54
             TIME(I)=3600*24*TIME(I)
             TR(I)=TIME(I)
             TR(I)=TR(I)/7
      20 CONTINUE
         XKAVG=0.
         XLTOTAL=0.
         DO 100 I=1,NLAYER
            XKAVG=XKAVG+1/XK(I)
            XLTOTAL=XTOTAL=XL(I)
     100 CONTINUE
         XKAVG=1/XKAVG
         H=0.
         TTR=0.
         TESTTIME=0.
         DO 200 I=1,NCYCLE
            DT=TR(I)
            H=H+HEAD(I)
            IF(TTR.LE.TFSS) THEN
            QSS = XKSS*EN
            Q=QSS+XKAVG*H/XLTOTAL
        ELSE
         Q=XKAVG*H/XLTOTAL
      ENDIF
         QFLOW=Q*AREA/(12*2.54)
      WRITE(6,*)TESTTIME,QFLOW,H/(12*2.54)
      DO 210 J=1,7
         TTR=TTR+DT
         DH=ABS(XKAVG*H*DT/XLTOTAL)
         H=H-DH
         IF (TTR.LT.TFSS) THEN
            QSS=XKSS*EN
            Q=QSS+XKAVG*H/XLTOTAL
         ELSE
            Q=XKAVE*H/XLTOTAL
         ENDIF
         TESTTIME=TTR/(3600*24)
         QFLOW=Q*AREA/(12*2.54)
         WRITE(6,*)TESTTIME,QFLOW,H/(12*2.54)
210  CONTINUE
200 CONTINUE
    TTOTAL=TTR/(3600*24)
    WRITE(6,5)QFLOW,TTOTAL
  5 FORMAT(//,' THE GROUNDWATER FLOW RATE = ',E10.2,'CU-FT./SEC',
   1         ,' IN ELAPSED TIME = ',F8.3,' DAYS')
    STOP
    END
```